.

TUNDRA VEGETATION RECOVERY ON 30 YEAR-OLD SEEDED AND UNSEEDED DRILLING MUD SUMPS IN THE MACKENZIE RIVER DELTA REGION, NWT.

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Abstract

Oil and gas exploration conducted in the 1970s left behind a legacy of abandoned well sites in the Mackenzie Delta region of northern Canada, including several in the Kendall Island Migratory Bird Sanctuary and surrounding areas. Evidence of 30 year-old well sites is present in the form of drilling mud sumps, which are mounds of disturbed tundra that contain frozen drilling-wastes. One to two years after the wells were decommissioned some of the sites were seeded with non-native grass species and fertilized to test whether these treatments could accelerate vegetation recovery and prevent erosion. The main objective of this research was to examine the long-term impact of post-disturbance seeding treatments on the vegetation recovery of drilling mud sumps.

Surveys of vegetation composition and environmental conditions at 12 sump sites (6 seeded and 6 unseeded) showed that, after over 30 years of recovery, seeded sumps in the Mackenzie Delta did not significantly differ from those left for natural recovery. However, seeded and previously introduced grasses *Festuca rubra* and *Poa pratensis* were found on both seeded and unseeded sumps. The undisturbed surrounding tundra seems to be resistant to invasion by these introduced grasses. However, these species could become invasive in the future, particularly in the context of warming in the North and increasing anthropogenic disturbance. The results of this study contribute valuable information on the long-term effects of revegetation treatments that is critical for making informed management decisions about the rehabilitation of industrial disturbances in the Arctic.

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List of Abbreviations

ANOVA: Analysis of Variance

DGPS: Differential Global Positioning System

GAM: Generalized Additive Model

GPS: Global Positioning System

KIBS: Kendall Island Migratory Bird Sanctuary

LSD: Least Significant Difference

MANOVA: Multivariate Analysis of Variance

MGP: Mackenzie Gas Project

MRD: Mackenzie River Delta

MRPP: Multi-Response Permutation Procedure

NRC: Natural Resources Canada

NWT: Northwest Territories

1. Introduction

1.1 Oil and Gas Exploration in the Mackenzie River Delta Area

Oil and gas exploration practices in the Mackenzie River Delta (MRD) region, Northwest Territories (NWT) were very active in the 1970s and resulted in the formation of numerous exploratory well sites and disturbed vegetation. In 1974 a northern pipeline, the Mackenzie Valley Pipeline, was proposed to extend from Alaska through the Mackenzie valley south to Alberta (Bliss 1983). However, there was much public debate over the pipeline, so the Government of Canada appointed Justice Thomas Berger to do a public inquiry into the project. Exploration activities ceased because of a ten year moratorium after the Berger inquiry in order to settle land claims in the region (Berger 1977).

In his report, Justice Berger stated that building the pipeline would cause too much damage to the environment due to summer construction since it would likely take more than one year to complete (Berger 1977). Now, the renewal of the Mackenzie Gas Project (MGP) is expected to result in industrial development in the North in the coming decade (2010s). The MGP is a proposed 1220 km buried natural gas pipeline that would connect three production facilities (Taglu, Niglintgak, and Parson's Lake) in the MRD to northwest Alberta (Mackenzie Gas Project 2003).

Understanding of the vegetation recovery from past oil and gas disturbances is critical for predicting the impacts of planned future development this region. In particular, information on the long-term effects of rehabilitation treatments on plant community recovery can provide valuable information to develop suitable rehabilitation efforts for oil and gas exploration in the MRD region.

There are several types of disturbances associated with oil and gas exploration (Truett and Johnson 2000). A disturbance can be defined as a sudden deviation from the normal or reference state, which in this case is considered the undisturbed surrounding tundra (Forbes et al. 2001). The focus of this thesis is specifically on surface disturbances caused by the removal, mixing, and re-deposition of plant material, soil, and permafrost materials during the formation and capping of drilling mud sumps.

A drilling mud sump refers to a large, below ground pit that was blasted (using dynamite) out of the permafrost, usually adjacent to the drilling well head. The drilling mud sumps were approximately 50 m x 25 m x 5 m deep depending on the depth of the well drilled (French 1980). During drilling, a mixture of salt (KCl), water, diesel fuel, mineral oil, and other chemicals were combined into a drilling mud, which was poured down the well to aid the drilling process and removal of the rock cuttings. The drilling-waste (drilling mud and rock cuttings) is toxic and therefore should not be released into the environment (AMEC Earth and Environment 2004). The drilling-waste was disposed of in the drilling mud sump. For example, a well that was 3000 m deep would have produced approximately 4000 m^3 of drilling-waste (AMEC Earth and Environment 2004).

When a well was decommissioned, the drilling-waste was usually allowed to freeze, and was then capped with the excavated permafrost materials (AMEC Earth and Environment 2004). This mound on the landscape is called a capped sump, referred to hereafter as a sump (Serverson-Baker 2004). In the NWT, it is standard to dispose of the drilling-wastes in these drilling mud sumps under the Territorial Arctic Land Use Regulations (French 1980). Sumps were intended to keep drilling-waste frozen in the permafrost and for the active layer (seasonal thaw layer) to recover to that of the surrounding area (Kokelj and GeoNorth 2002). If the drilling well, and thus the sump, was kept open in the summer the permafrost in the sump could degrade and create an even larger sump (Kokelj and GeoNorth 2002).

There are two major concerns with disposing of drilling-wastes in sumps: leakage of drilling-waste and permafrost degradation. Leakage of drilling-wastes, in particular salts, kills the vegetation and contaminates the soil (Smith and James 1985, McKendrick 1991). Annual flooding and storm surges in the low-lying areas of the MRD increase the potential for the drilling-waste to come in contact with flood waters and cause environmental degradation (AMEC Earth and Environment 2004). Permafrost degradation can result in the leakage of drilling-wastes into the surrounding tundra. The flooding has also resulted in ponding around sumps, which can cause warming of the underlying permafrost and thermokarst erosion (AMEC Earth and Environment 2004). Thermokarst is the melting of ice-rich permafrost (Mackay 1970). Overall, sump success has been limited in the MRD. Rehabilitation of plant cover on sumps may help to prevent leakage of drilling-waste and thermokarst.

1.2 Low Arctic Vegetation and Disturbance Recovery

The tundra biome makes up 15 % of the world's land surface (Bliss and Wein 1972). Low species diversity, simple community structure, and low annual productivity are all characteristics typical of arctic tundra communities (Forbes et al. 2001).The low arctic zone of the tundra biome refers to areas with a high percent cover of vascular plants located north of the latitudinal tree line (Bliss 1983). The growing season in the low arctic tundra is typically three to four months, which allows little time for plant growth (Forbes et al. 2001). Vegetation growth during this short period is limited by nutrient availability, especially nitrogen and phosphorus, soil moisture, and temperature (Bliss 1988). To survive in this region, species must be both morphologically and physiologically adapted to the severe habitat conditions and able to rapidly complete annual life history cycles of growth and reproduction (Svoboda and Henry 1987). Due to these restrictive conditions, spread of invasive plant species in the low arctic has been historically low (Canadian Food Inspection Agency 2008).

The low arctic tundra of the MRD is composed of vegetation dominated by low-lying shrubs, grasses, and sedges, and is underlain by continuous permafrost (Mackay 1963). Natural disturbances in the MRD region include flooding and river erosion, the formation of ice wedges and polygonal tundra (Bliss and Peterson 1992), and thaw slumps caused by permafrost degradation (Walker and Walker 1991). Because the MRD is an active delta influenced by both the Mackenzie River and the Beaufort Sea, the lowland areas are subject to flooding in the spring and during storm surges (Mackay 1963).

Recovery of tundra vegetation from disturbance is complex and depends on many factors that are closely associated with the physical environment (Walker and Walker 1991, Truett and Kertell 1992). These factors include the presence of a buried seed bank, the dispersal of seeds from surrounding areas, competitive interactions, underlying substrate, and the moisture, temperature and biogeochemical regime (Walker and Walker 1991, Forbes et al. 2001). The sumps, which are the focus of this study, have little to no buried seed bank because they were capped with excavated permafrost materials. Thus, plant colonization of sumps largely relies on seed dispersal from the surrounding tundra (Forbes et al. 2001). Vegetation is necessary to stabilize the soil and prevent erosion, and to maintain good quality habitat for wildlife (Haag and Bliss 1974, Shirazi et al. 1998). To minimize the impacts of human development, it is critical to study the long-term effects of disturbances to develop management strategies that maximize the potential for ecosystem recovery.

Recent work in the MRD has demonstrated that vegetation communities on sumps were different from the undisturbed surrounding tundra more than 30 years after disturbance (Johnstone and Kokelj 2008). Naturally revegetated sumps had a distinct plant community dominated by grasses and herbaceous species, which were present in low abundance in the undisturbed surrounding tundra. Differences in vegetation were found to be strongly associated with elevation and moisture gradients suggesting that the impacts of sump disturbances persist due to effects on microtopography and associated drainage. Johnstone and Kokelj (2008) also found an association between tall shrubs and increased snow depth on the sumps, which may lead to increased soil temperatures and subsequent permafrost degradation on the sumps. These findings suggest 30 years is not enough time for vegetation communities to recover to their undisturbed composition after the construction of sumps in the MRD. However, it is possible that reclamation treatments may modify the rate or pathway of recovery from disturbance.

Given the slow natural recovery of tundra vegetation from disturbances, there has been substantial interest in developing strategies to accelerate vegetation recovery following industrial disturbances. With natural revegetation, the total plant cover may range from only 30 to 60 % after six years (Hernandez 1973). Therefore, different revegetation strategies have been tried in order to speed up recovery of disturbances in the tundra. Arctic revegetation research began in Alaska and Canada in the 1970s; however the most extensive revegetation studies are from Alaska in the Prudhoe Bay and Kuparuk oilfields on the Alaska Coastal Plain. Many techniques for rehabilitation have been tested on gravel pads/pits, pipelines, roads, and oil spills. In the past, revegetation treatments have been used in order to prevent erosion (Younkin and Martens 1976), provide canopy cover (Forbes and Jefferies 1999), reduce the visual impact of scars, and enhance habitat quality for wildlife (Truett and Kertell 1992). The initial goals of many revegetation studies were to facilitate seedling establishment and to accelerate plant growth (biomass, cover) in the short-term. Now, revegetation of disturbed sites is also aimed at preventing thermokarst and facilitating the colonization of indigenous plant species (Kidd et al. 2006).

Revegetation treatments can include fertilizer applications in combination with a seeding treatment using both cultivated and indigenous species. Assisted revegetation involves site preparation and monitoring and it can also include surface modification, fertilization, soil amendments, and light seeding (Wright 2008). Fertilizer treatment usually accompanies seeding as it increases productivity and the likelihood of plant establishment because nutrients, especially phosphorus, are limiting in this environment (Bliss and Wein 1972). Fertilizer treatment has a positive effect on indigenous species colonization (Younkin and Martens 1976) and aids in the establishment of primary successional species, i.e. sedges and grasses, of disturbed areas (Forbes et al. 2001). Re-fertilization in the second year has also been found to enhance seeded plant growth in the tundra (Hernandez 1973, Younkin and Martens 1976). Seeding treatments can be done with seed from commercially available non-native species, native species cultivars, or indigenous species. Non-native species are species that are not found in the undisturbed "native" tundra of that region. Native species cultivars are plant species that are found in the undisturbed tundra that were harvested and then commercially produced for seeding. Indigenous species are species that are harvested on-site as seeds, cuttings, or plugs from the undisturbed surrounding tundra. Below I discuss the issues that have been associated with these seeding treatments into natural tundra habitats.

In the short term, seeding with non-native species can aid in erosion control on disturbed sites. In the MRD, Younkin and Martens (1976) found that three years after seeding sumps with non-native species, initial litter cover provided aid in erosion prevention. The seeded grass species *Festuca rubra* cultivar "Boreal" and *Poa pratensis* cultivar "Nugget" consistently survived at least two winters and showed aggressive spread when seeded in monocultures (Younkin and Martens 1976). Other seeded grass species died back after the first winter and provided beneficial litter cover (Younkin and Martens 1976). The total cover provided by seeded species after the second year was less than 13.7 % (Younkin and Martens 1976). They also found that indigenous species cover was inversely proportional to seeded species cover and consistently less in seeded compared to unseeded sites. Non-native grass species have been shown to outcompete with indigenous species for nutrients (Evans and Kershaw 1989). Some studies have suggested that non-native species can delay or inhibit the colonization of disturbed areas by indigenous species (Younkin and Martens 1987, Jorgenson 1997, Jorgenson et al. 2003, Kidd et al. 2006). For example, indigenous species cover, dominated by *Arctagrostis latifolia* (R. Br.) Griseb. (Wideleaf Polargrass), and *Carex bigelowii* Torr. ex Schwein. (Bigelow's Sedge), was inhibited by seeded grasses and ranged in cover from 7-80% cover after twelve years (Younkin and Martens 1987). Many studies often suggest alternatives to seeding with non-native species because of these issues.

In the low arctic, native species cultivars and varieties have been used in revegetation treatments because they are the natural colonizers after disturbance, they are suited for the harsh arctic environment, and they are available commercially (Kershaw and Kershaw 1987). Jorgenson et al. (2003) found that seeding with native species had a similar recovery rate as seeding with non-native species; the plant cover was similar to adjacent undisturbed wet tundra after 15-25 years. In Alaska, three grass species, *Poa glauca* variety "Tundra," *Arctagrostis latifolia* variety "Alyeska," and *Festuca rubra* variety "Arctared" have been identified for seeding because they are adapted to the arctic and can be produced for seed outside the arctic

(McKendrick 1991). These and other similar cultivars are often used in revegetation trials in Canada (e.g. Younkin and Martens 1976); however, in many locations in Canada they would be considered non-native species.

More recently, indigenous species have been used for revegetation purposes as cuttings (e.g. willows), transplants/plugs, and nurse plants and have proven effective because they provide high species richness (Everett et al. 1985, Kershaw and Kershaw 1987, McKendrick 1987, Shirazi et al. 1998, Jorgenson et al. 2003, Kidd et al. 2006). Indigenous species are collected from adjacent undisturbed tundra because they are not available commercially. Collected species have included shrubs such as *Salix* spp., forbs including *Artemesia* spp., and graminoids like *Deschampsia caespitosa* (L.) Beauv. (McKendrick 2000). The purpose of using transplants or cuttings is to create islands of tundra vegetation that reproduce asexually to increase cover as well as providing a seed source on disturbances (Kidd et al. 2006). Kidd et al. (2006) found that seeding with indigenous graminoids species and fertilizer was comparable to transplants of indigenous species and fertilizer, but seeding lead to a slightly higher richness after 13 years. Increased species richness may give plant communities a greater ability to adapt to the changing environment and aid in recovery (Kidd et al. 2006). In the subarctic of northern Quebec, revegetation of disturbed ATV tracks in village courtyards using indigenous forb species and fertilizer resulted in the establishment and growth of seeded species within two growing seasons (Deshaies et al. 2009). There have also been successful short-term (1-2 years) revegetation experiments in Churchill Manitoba using several indigenous legumes, forbs, and grasses to restore gravel pits (Rausch and Kershaw 2007). Nurse plants such as mosses can also be transplanted to create safe sites i.e. a favourable environment for seed germination and establishment, which facilitate indigenous species colonization and organic mat formation

(Forbes 1993, Forbes and Jefferies 1999). However, when using indigenous species, one must be cautious and use non-destructive collection methods when possible in order to protect the donor sites (Urbanska 1997).

Seeding has been done mostly with graminoids, but over half of the colonizers of disturbed areas are forbs (Kershaw and Kershaw 1987, Jorgenson and Joyce 1994, Forbes and Jefferies 1999, Kidd et al. 2006). Seeding with a variety of plant functional types and species may help develop a diverse plant community that is more suitable for wildlife (McKendrick 1991). Leguminous forbs such as *Astragalus alpinus* L. and *Hedysarum alpinum* L. fix nitrogen, which is a limiting nutrient in the arctic, and they are one of the first colonizers to gravel soils (Forbes and Jefferies 1999). Shrubs such as *Salix* spp. are fast growing, produce a large quantity of seeds, and are long lived (Bliss and Cantlon 1957), thus providing long-term vegetation cover. Seeding with indigenous grasses such as *Puccinellia langeana* (Berlin) Sorensen ex Hultén (McKendrick 1997) and legumes has been successful in Alaska (Johnson 1981, Jorgenson et al. 2003, Alaska Biological Research Inc. 2006).

When considering revegetation on sumps, especially those that have leaked drilling-wastes, seeding with salt-tolerant plants is a viable option. Indigenous plant species such as *Puccinellia langeana, P. arctica* (Hook.) Fern. & Weatherby, *Arctophila fulva* (Trin.) Rupr. ex Anderss., *Dupontia fisheri* R. Br., and *Senecio congestus* (R. Br.) DC. are able to colonize saline soils and are palatable to wildlife (McKendrick 1996, McKendrick 2000). Along with seeding, the removal of sodium, which is the cause of salt damage to soils and drought in plants, can be accomplished by adding chemicals such as calcium nitrate to the soil (McKendrick 1996). Restoring the insulating organic layer is also important in order to reduce evaporation and reduce the concentration of salts in the soil (Forbes and Jefferies 1999). This may be done by adding organic mulch to affected areas and then seeding or transplanting the appropriate species. Some authors suggest that areas that are not threatened with erosion problems should not be seeded because indigenous vegetation is better adapted to the cool environment, and low nutrient content soils (Younkin and Martens 1976, Chapin and Chapin 1980, Cargill and Chapin III 1987). Caution must be exercised when seeding with non-native species because they may persist and spread in the ecosystem (Hobbs and Huenneke 1992).

1.3 Research Gaps

There are several gaps in our understanding of revegetation treatments in the Canadian Arctic. Because few long-term studies have been conducted, it is unknown what effect seeding treatments have on plant community recovery over several decades. In particular, there have been no multi-decadal studies on the effect of seeding treatments on vegetation recovery from disturbance in the western Canadian arctic. Such studies are needed to address whether seeded plant species create a barrier to indigenous plant species recovery in the long-term

Another issue that must be addressed is the potential for seeded non-native species to invade the undisturbed surrounding tundra. One study of abandoned rig sites in the Caribou Hills, NWT found that after twelve years, seeded non-native species survived and provided 20-60% cover in seeded plots and also invaded adjacent unseeded (control) plots with 5-60% cover (Younkin and Martens 1987). This study suggests that there is the possibility for seeded species to invade the undisturbed tundra. In general, there is a lack of knowledge on the invasion of the arctic tundra by introduced species. Introduced species have been naturalized in areas to which they were not native and may maintain themselves in waste places or natural disturbances (Hernandez 1973). These introduced species may be considered invasive if they decrease or replace native plant and animal species or alter ecosystem functioning (Hobbs and Huenneke

1992). Disturbance has been shown to facilitate species invasion with negative impacts in many ecosystems (Hobbs and Huenneke 1992). In Russia, introduced boreal and subarctic species *Polygonum humifusum* Pallas and *Rorippa palustris* (L.) Bess. have migrated north along roads and corridors built for pipeline construction and have spread into disturbed areas where they out-compete native species (Forbes 1997). Additional research is needed to assess whether seeding treatments of disturbances such as sumps may facilitate the invasive spread of non-native species.

Impacts of disturbance and revegetation must also be considered in the context of climate change in the North. The climate is warming in the Arctic and human disturbances may increase the impact of climate change in this region (Forbes et al. 2001, Anisimov et al. 2007, Huntington et al. 2007, Burn and Kokelj 2009). If climate change causes an expansion of suitable habitat in the Arctic, the possibility of introduced species becoming invasive may be increased (Tape et al. 2006).

. Lastly, the cumulative impact of disturbances and revegetation treatments from past industrial activities has not been determined for many regions. In order to assess the cumulative impact of disturbance on the arctic tundra we need to be able to understand past disturbances and their recovery as well as be able to predict future impacts. For example, the Kendall Island Migratory Bird Sanctuary (KIBS) is located in the outer Mackenzie Delta and is home to numerous migratory and nesting birds protected by the Migratory Birds Convention Act (Parks Canada 1992). This area contains large natural gas reserves and is the proposed location for two gas production facilities for the MGP. There is a government mandate that less than 1 % ($3.35 km^2$) of the total land in the KIBS can be impacted by anthropogenic disturbances, including both old and new impacts (Canadian Wildlife Service 2004). It has been estimated that

2.5 % of the terrestrial habitat in KIBS has been disturbed in the past by seismic lines, drill pads, two staging areas, a permanent camp, and an airstrip, but their current recovery is not always known (Ashenhurst 2004). The footprint of the sumps in KIBS is not known, but if most sumps have not yet recovered (Johnstone and Kokelj 2008) they should be included as part of the current assessment of cumulative disturbance impact in the sanctuary.

1.4 Study Objectives

The objectives of this study are to a) determine whether early seeding and fertilizer treatments applied to sump disturbances have a detectable effect on long-term vegetation recovery, and b) gather more information on the decadal-scale effects of disturbances associated with oil and gas exploration on tundra plant communities in the Mackenzie Delta. Tests of seeding effects are based on comparison of sumps that were seeded with non-native species and fertilized within 1-2 years after disturbance (Younkin and Martens 1976) to those left for natural recovery (unseeded). Because these seeding treatments applied a mix of five non-native grass species, I hypothesize that, in particular, some seeded and non-native grass species may have persisted as naturalized populations on the seeded sumps or expanded into surrounding undisturbed tundra. General effects of sump disturbances on the plant community and environment are examined across two terrain types, with the aim of gaining more information on the long-term impacts of disturbance patterns between lowland and upland terrain. The results of this study are important for assessing the cumulative impacts of industrial disturbance in arctic environments, and provide managers with information on the long-term effects of revegetation and disturbance on plant communities in the MRD region.

2. Methods

2.1 Study Area

The study area is located in the outer Mackenzie Delta, NWT between 68°88'03" N and 69°38'89" N, and 133°29' W and 135°34' W in the Kendall Island Migratory Bird Sanctuary (KIBS), and in adjacent upland areas near Parson's Lake (Figure 2.1). Two of the proposed gas production facilities for the Mackenzie Gas Project are located in KIBS at Taglu Island and Niglintgak Island, and the third is located in the uplands near Parson's Lake.

The mean growing season for the low arctic tundra is three to four months, with a mean daily July temperature at Inuvik of 14.2° C and mean daily January temperature of -27.6° C (Forbes et al. 2001, Environment Canada 2009). In the winter there is continuous darkness and in the summer continuous light (Bliss 1988). The precipitation is low and mostly occurs as rain in July and August (Mackay 1963). The average dates for ice break-up and freeze-up of the Mackenzie River near Reindeer Station, ~50 km north of Inuvik, are 27 May and 18 October respectively (Mackay 1963).

The vegetation community of the lowland area is typically wet sedge/shrub meadows (Johnstone and Kokelj 2008). The vegetation characteristic of a wet sedge/shrub meadow has little lichen cover, abundant mosses, sedges including *Carex aquatilis* Wahlenb. (Water Sedge), *Eriophorum angustifolium* (Tall Cottongrass), and *E. scheuchzeri* Hoppe (White Cottongrass), grasses such as *Arctagrostis latifolia* (R. Br.) Griseb. (Wideleaf Polargrass), *Dupontia fisheri* R. Br. (Fisher's Tundragrass), and *Arctophila fulva* (Trin.) Rupr. ex Anderss. (Pendantgrass), and shrubs such as *Salix* spp. (Willow) (Bliss and Matveyeva 1992, Johnstone and Kokelj 2008). The lowland is frequently flooded and soils are alluvial in origin (Mackay 1963, Morse et al. 2009).

The upland terrain is characterized by glaciofluvial soils that are dry in the summer (Bliss and Wein 1972). These areas are well-drained and dominated by shrub-heath tundra (Bliss and Wein 1972, Johnstone and Kokelj 2008). Shrub-heath tundra is dominated by slow growing deciduous or evergreen shrubs e.g. *Arctostaphylos rubra* (Rehd. Wilson) Fern. (Red Fruit Bearberry), *Cassiope tetragona* (L.) D. Don (White Arctic Mountain Heather), *Empetrum nigrum* L. (Black Crowberry), *Ledum palustre* L. subsp. *decumbens* (Ait.) Hultén (Marsh Labrador Tea), *Rhododendron lapponicum* (L.) Wahlenb. (Lapland Rosebay), *Vaccinium uliginosum* L. (Bog Blueberry), and *Vaccinium vitis-idaea* L. (Lingonberry; Bliss and Wein 1972). These species are extremely sensitive to physical damage by disturbance, as even a single pass of a vehicle can be lethal for an individual (Bliss and Wein 1972, Hernandez 1973, Forbes et al. 2001).

2.2 Study Design

A revegetation experiment aimed at testing the effects of seeding and fertilization treatments on post-disturbance recovery was applied to a set of 22 sump sites in 1974-75 (Younkin and Martens 1976). Seeding treatment used several non-native grass species including *Festuca rubra* L. cultivar “Boreal” (Boreal Creeping Red Fescue), *Poa pratensis* L. cultivar “Nugget” (Nugget Kentucky Bluegrass), *Phleum pretense* L. (Climax timothy or Engmo timothy), *Phalaris arundinacea* L. (Frontier Reed Canary Grass), and *Lolium perenne* L. (Prolific Spring Rye) in different mixes. These sites were also aerially fertilized with a 14-28-14 mixture of $N\text{-}P_2O_5\text{-}K_2O$ (nitrogen-phosphate-potash) in the spring (Younkin and Martens 1976). Seeded sites were monitored for survival and percent cover of these species for three years (Younkin and Martens 1976). *Poa pratensis* and *F. rubra* were introduced into the MRD prior to the seeding trials and occur along road sides and near settlements (Porsid and Cody 1980). There were no reliable morphological features to separate the seeded cultivar and previously introduced varieties of these species; therefore the seeded and introduced individuals could not be distinguished.

This study was designed to compare the post disturbance recovery of vegetation on sumps with and without a revegetation treatment. Decommissioned sumps at KIBS and Parson's Lake were selected for study in 2008 based on four criteria: 1) sump decommission dates in the period 1972-1977, 2) proximity to the river channel (<1 km) for access to KIBS sites, 3) location of sumps on crown land to provide permission of access, 4) and for seeded sites, seed applications by Younkin and Martens (1976).

Thirteen sump sites (7 unseeded and 6 seeded) were sampled in total (9 lowland sites and 4 upland sites). Many of the sites seeded by Younkin and Martens (1976) were outside the study area or beyond our access range from the river channel. Consequently, our sample represents the maximum number of seeded sites that we were able to access by boat within KIBS, plus one pair of upland sites at Parson's Lake that were accessed with limited helicopter support from Indian and Northern Affairs Canada. One site at KIBS, B-19 (unseeded lowland) was sampled but not included in subsequent data analysis because it was located on naturally saline mud flats and supported plant communities that were distinct from all other sites in the study (AMEC Earth and Environment 2004). As a result, the data reported here are based on a total sample of 6 seeded and 6 unseeded sumps.

Although sump sites were unlikely to have been randomly selected *per se*, their distribution in the outer Mackenzie Delta suggests that they were selected without bias from available terrestrial habitats during well exploration in the 1970s. The sump sites were chosen by industry based on several factors including proximity to the well head and on ground with higher elevation to prevent ponding and minimize leakage of drilling-waste or collapse of the sump into a river channel (AMEC Earth and Environment 2004). The aerial seeding and fertilization experiment of Younkin and Martens (1976) was applied to the entire sump at the treated sites used in this investigation.

2.3 Field Data Collection

Field data collection occurred between 15 July and 15 August 2008. Surveys of vegetation and environmental conditions were conducted along two linear transects at each site. Transects ranged from 100-200 m in length depending on the shape of the sump and were divided into three zones representing different levels of disturbance associated with a sump: cap, perimeter, and surrounding undisturbed tundra (Figure 2.2 and 2.3). These zones were determined visually based on elevation, active layer depth, and indicators of disturbance including pilings and refuse left from exploration activities. The cap zone consisted of an elevated disturbed area created from soil overburden piled on the top of the sump, and the perimeter zone was a flat or concave transitional zone between the cap and the surrounding undisturbed vegetation (Figure 2.4). Transects started from the center of the cap and went into the surrounding undisturbed tundra. The heading of the first transect was randomly selected and then the second transect was placed perpendicular to the first from the center of the sump. Where the sumps were relatively small or collapsing transects crossed the diameter of the sump cap to increase the length of the cap zone in order to capture relatively equal transect lengths for each zone. If either the first or second transect crossed standing water that was too deep to walk through with chest waders, a new heading was randomly selected.

Plant community composition was measured along the transects using a stratified random design. Presence/absence data for all vascular plant species encountered were collected in six 0.5 by 0.5 m quadrats per zone, randomly placed along the transect within that zone (Figure 2.3). Presence/absence was used instead of percent cover to measure plant community composition because it is less subjective and more efficient, allowing for the sampling of more plots (Green 1979, Hirst and Jackson 2007). Visual percent cover was estimated in each quadrat for several surface cover variables: total vegetation, lichen, litter, water, bare soil, and moss. Total vegetation cover was measured in order to determine an aggregate community response. Litter was defined as any fallen, unattached dead vegetation. The maximum height of the vegetation canopy in each quadrat was also measured from the ground to the tallest plant using a meter stick. Easily recognized species were identified in the field according to both Porsid and Cody (1980) and Cody (2000) and voucher samples were collected for species of uncertain identity.

Plant voucher specimens were taken to the University of Saskatchewan for identification. Species were identified following the Flora of the Yukon Territory (Cody 2000), Vascular Plants of Continental Northwest Territories, Canada (Porsid and Cody 1980), or Budd's Flora of the Canadian Prairie Provinces (Looman and Best 1979). The nomenclature followed the Integrated Taxonomic Information System online database (ITIS 2009). Some species were not identified because of lack of proper structures (flowers, fruit), missing specimens, or lack of comparative specimens in the W.P. Fraser Herbarium (SASK) of the University of Saskatchewan. These specimens (4 Cyperaceae, 7 Poaceae, 4 Salicaceae, and 9 unknown herbs) were not included in subsequent analyses. The voucher specimens will be kept with Dr. Johnstone in the Northern Plant Ecology Lab, University of Saskatchewan.

Environmental conditions were assessed in the field by taking soil cores and measuring organic soil depth, elevation, and active layer depth (Figure 2.4). Soil conductivity was also measured on soil samples in the lab. Soil cores, using a 5 cm diameter soil corer, were collected in three randomly selected quadrats per zone along each transect (n = 18/site). The organic layer (horizon) depth was measured from the core sample and included all the organic material between the first mineral soil horizon and the base of the surface litter or live moss. A sample of the mineral soil was collected from 5 cm below the surface of the mineral soil. The samples were stored in plastic bags and shipped to Yellowknife at the end of the field season. The samples were analyzed for electrical conductivity by a technician using an EC meter at the Taiga Environmental Laboratory, Yellowknife, NWT. Electrical conductivity was used as an indicator of soil salinity (Corwin 2003). High soil salinity can cause reduced plant growth, reduced biomass, or plant death (Corwin and Lesch 2003). Non-saline soils generally have a conductivity of less than 2 decisiemens per meter (dS/m) whereas saline soils can range from weak saline (2-4 dS/m), to severely saline (>20 dS/m) (Dunn 2001). Active layer depth was measured every 10 m along the transect by pushing a 120 cm calibrated steel probe into the soil until the depth of refusal. In order to prevent bias associated with the variation in active layer thickness over the sampling period, the order of sampling of seeded and unseeded sites was randomly determined.

Elevation data were collected every 5 m along a transect using a Trimble R3 differential GPS system (L1 GPS receiver, A3 L1 GPS antenna). The data were processed using the Trimble software version 1.11 (Trimble Navigation Limited 2007) and then plotted and examined for inconsistencies. Some sites had a number of inconsistencies where, for example, a transect showed sudden valleys and peaks where the ground was actually relatively flat. To correct these errors, the processed Trimble points (decimal degree coordinates) were exported to an online global GPS post-processing service (Natural Resources Canada), which processed the raw Trimble points into more accurately estimated points (NRC points). The original points from the Trimble software had a single value for each 5 m point along the transect but output from the NRC post-processing were raw GPS readings with 20-30 points for each Trimble point. To extract the original transect points, the NRC points were imported into the statistical program R version 2.8.0 (R Development Core Team 2009) and fit using a general additive model (Wood 2006). Latitude and longitude were smoothed variables using the GAM function in the mgcv package of R (Wood 2008). Latitude and longitude were allowed to interact in the GAM. This was done in order to effectively fit the actual surface of the site. Each site had a sufficient number of points (2000-3000) so that over-fitting the surface was not a concern. Visual inspection of plots of the fitted data versus to the original Trimble points confirmed that the fitted points were equivalent to the original data, minus the original observed irregularities. The GAM fitted elevation points were averaged for the nearest quadrat and then standardized by the lowest point for each site to obtain the relative elevation of each quadrat at a site.

2.4 Statistical Analysis

2.4.1 Vegetation Patterns

For analyses of vegetation composition, the 12 quadrats (6 per transect) within a zone were summed at each site and then divided by 12 to get a frequency. I used quadrat frequencies because individual quadrats were small (0.25 m^2) and were therefore unable to fully represent plant community composition within a disturbance zone at a site.

I used PC-ORD version 5.19 (McCune and Mefford 2006) to perform non-metric multidimensional scaling (NMS) to ordinate the sites in species space (Kruskal 1964). NMS is robust to non-linear relationships and produces interpretable results when analyzing ecological data (McCune and Grace 2002). NMS also avoids the assumption of normality. This analysis technique uses ranked distances, which improves its ability to extract the dominant pattern using non-metric data (McCune and Grace 2002). It is an iterative solution and will produce the best positions of the sample plots in species space to reduce stress (McCune and Grace 2002). I chose the Sørensen distance measure (Bray-Curtis) because it has been shown to be effective for ecological data (Faith et al. 1987). The ordinations were based on species proportions and a secondary matrix of environmental variables and surface cover were used as overlays on the species ordination. The environmental variables assessed were frequency values per zone of relative elevation (m), active layer depth (cm), conductivity (dS/m), and organic layer depth (cm) at each site. Surface cover variables were averaged measures of cover for total vegetation, lichen, litter, water, bare soil, and moss.

An outlier analysis was conducted using PC-ORD on the species matrixes for both lowland and upland data sets. An outlier was defined as a species with an average Sørensen distance of greater than 2.5 standard deviations from the other pairs of species. Species present less than twice were deleted. Three species were detected and removed from the data set: *Deschampsia caespitosa* (L.) Beauv., *Festuca richardsonii* Hook., and *Salix reticulata* L.. These species were only found in a single site and were removed because it created a better NMS plot with less stress. McCune and Grace (2002) reported that removing rare species can reduce noise in ordinations without losing much information. This approach has been employed in other studies (Clarke et al. 2006). Site D-58 (undisturbed zone) was also detected as an outlier, but it was not deleted because it was essential to interpretation of the data set.

I ran all NMS ordinations using a Sørensen distance measure and the autopilot function with a random starting configuration and 200 independent, iterative runs with real data. The number of ordination axes to use in the final solution was determined using a Monte Carlo test with 50 runs. Rank correlations (Kendall's tau) between environmental variables and ordination axes were used to assess similarities between sites and zones of disturbance. The variables with stronger correlations ($\tau > 0.200$) were used as vector overlays in the ordination illustrations.

Three separate NMS ordinations analyzed based on 1) combined lowland and upland terrain data, 2) lowland terrain data only, and 3) upland terrain data only. I also carried out an MRPP (Multi-Response Permutation Procedure) with terrain as a grouping variable to determine if there was a difference in species composition between lowland and upland sites (Mielke and Berry 2007). MRPP is a non-parametric test that is used to determine the difference between two or more *a priori* groups (McCune and Grace 2002). I used MRPP because it does not require the data to be multivariate normal or have equal variances (McCune and Grace 2002).

Indicator species analysis (Dufrene and Legendre 1997) was used in order to determine which species were indicators of an *a priori* group. A perfect indicator of the group is one that is always present and is exclusive to that group (Dufrene and Legendre 1997). Indicator values were tested for significance using a Monte Carlo test in PC-ORD. The site-level data were used for indicator species analyses based on 1) combined lowland and upland data, 2) lowland data, and 3) upland data. The indicator value for a species was calculated by taking the relative abundance and frequency of each species in each group and using 4999 randomizations to identify significant indicator species. An indicator species has a high indicator value and a low probability ($p < 0.05$) of obtaining an indicator value of equal of higher value by chance

(McCune and Grace 2002). A significant indicator species is a species that is highly characteristic of that *a priori* group (Dufrene and Legendre 1997).

2.4.1.1 Lowland Terrain

A NMS ordination and an indicator species analysis were used to assess patterns of plant community composition in the lowland terrain data, as outlined above. NMS ordination was used to visualize patterns in the plant community data that may not have been apparent when both upland and lowland data were included in the analysis. I used an indicator species analysis to determine which species were significant indicators for the disturbance zones and seeding treatments. These data were also used for a hierarchical cluster analysis in order to determine if the unseeded and seeded sites clustered together based on the species composition distance matrix. I used a Flexible Beta (β) linkage of -0.25 and a Sørensen distance measure (McCune and Grace 2002).

Species diversity was measured using species richness and Simpson's index of diversity (Simpson 1949). Simpson's diversity (D) applies to an infinite population and measures the probability that two randomly chosen individuals will be the same species (Simpson 1949). Richness and Simpson's diversity was calculated using the row/column summary function in PC-ORD.

Subsequent data analysis for the lowland sites followed a mixed model, split-plot design (Quinn and Keough 2002). The zones of the sump were not chosen at random, which creates a restriction on the split-plot design used to analyze the lowland plant community. There were only four sites sampled in the upland terrain, which was insufficient to test for differences between lowland and upland data as well as testing for a seeding and zone interaction in the upland data alone. Despite the possible biases, the split-plot is a standard statistical design in agriculture and it is the best available method for testing if the seeding treatment had an effect on the plant community and if they varied between the zones. In this design, sites (n = 4 per treatment) were considered random factors (B(A)) nested within the seeding treatment (A). The seeding treatment was treated as a fixed factor with two levels, seeded and unseeded (Table 2.2). The within-subjects, or split-plot factor was the zones of the sump (C), which was also considered a fixed factor with three levels: cap, perimeter, and undisturbed control.

I used Analysis of Variance (ANOVA) with a split-plot design to test hypotheses of whether the seeding treatment had an effect on a given univariate response variable, and if the effects varied between disturbance zones (cap, perimeter, and undisturbed). Pair wise comparisons using least significant difference (LSD) were used to assess significant differences between the three zones (Zar 1999). Assumptions of equal variance and covariance of the groups where tested by determining if the data were spherical, which was one of the underlying assumptions of a split-plot ANOVAs (Zar 1999). I transformed the richness data using a $\log_{10}$ transformation in order to increase equality of variance. Simpson's diversity was not transformed because the data had equal variance across the mean. All data analyses were run in SPSS version 17.0 (SPSS Inc. 2009) unless otherwise stated.

To test for effects of seeding treatment or disturbance zones on multivariate community composition and surface cover, I applied a Multivariate Analysis of Variance (MANOVA) using a split-plot design. The response variables were 1) plant functional types and 2) surface cover variables. A MANOVA was chosen because it is more appropriate than multiple ANOVAs when the variables being tested are correlated (Zar 1999). Moreover, MANOVAs are more sensitive to group differences and thus if there was a treatment effect, the MANOVA would detect it (Zar 1999). I used the Pillai's trace test statistic because it is considered more robust to unequal covariance and best for general use (Pillai 1955, Zar 1999). If the results were significant, subsequent ANOVAs were performed to determine which variables were different followed by LSD pair wise comparisons, as described above.

Species were classified into functional types for MANOVA analyses because plants in a group may often play similar roles in the community and functional types may respond differently to disturbance (Forbes et al. 2001, McLaren 2006). I analyzed richness within a functional type because diversity within these groups is important for the maintenance of ecosystem functions in the face of disturbance or other perturbations (Chapin 1993, Chapin et al. 1996). This classification was considered to be more ecologically relevant than alternative classification schemes based on species life history such as reproductive strategies, physiological adaptations such as phenological development, or Braun-Blanquet classification systems that assigns species based on their fidelity to particular associations (Braun-Blanquet 1932, Bliss 1988). Plant functional types share several physiological adaptations related to growth rate including rates of photosynthesis and nutrient absorption, biomass accumulation, and transpiration rate (Chapin III 1993). Therefore, classifying plants according to functional type is a simplified way of grouping plants with both ecological and physiological importance.

Plant species were classified into six functional types: legumes, non-legume forbs, grasses, sedges/rushes, deciduous shrubs, and evergreen shrubs (Chapin et al. 1996). Legumes, made up of species from the Fabaceae family, were separated from the other herbaceous species because they play an important ecological role in fixing nitrogen (Chapin et al. 1996). Non-legume forbs were herbaceous species. Grasses were separated from sedges/rushes because they have been shown to colonize disturbances whereas sedges/rushes usually dominate wet areas (Chapin et al. 1996). Deciduous shrubs are more common in nutrient rich upland sites whereas evergreen shrubs dominate dry heath sites. In order to make the data more spherical, the plant functional type data were $\log_{10}$ (x+1) transformed. At the lowland sites, only one species of evergreen shrub, *Vaccinium vitis-idaea* L., was present at one site and this functional group was excluded from the analysis.

The MANOVA analysis of surface cover included percent cover of the following variables: litter, bare soil, moss, and total vegetation cover. Lichen and water cover were excluded from the analysis because data for these variables were largely zeros. The bare soil and moss cover data were $\log_{10}$ (x+1) transformed in order to make the data more spherical. The data were then normalized using Z-scores (Zar 1999).

Because the number of sample sites (n=8) was small relative to the number of response variables (≥4), the MANOVAs had insufficient residual degrees of freedom to provide a multivariate test statistic for the zones or the interaction between zones and treatment. SPSS dealt with this problem by automatically running a separate within-subjects test on averaged, standardized variables. I used the test statistic from this test to determine if zone had an effect. If the zone effect was significant, subsequent ANOVAs were used for each variable separately in order to determine which variables where different. LSD pair wise comparisons were used to determine the pattern of significant differences between zones.

2.4.1.2 Upland Terrain

NMS ordination and indicator species analyses were also applied to a separate analysis of the upland plant community data set. The analysis procedure followed the same protocol outlined in the previous section. Tests of seeding treatment and disturbance zone effects were carried out slightly differently for upland data due to the fact that the sample size was too small (n = 4 sites total) to run the analysis as a split-plot design. To test for a seeding treatment effect, one-way

ANOVA (species richness, diversity), or MANOVA (plant functional types, surface cover) tests were run using only the cap zone data. The cap zone data was chosen because that is where most of the variation was expected to occur among the seeded and unseeded sites. For analyses that indicated no seeding treatment effect, I applied subsequent ANOVAs or MANOVAs to test for a disturbance zone effect with the seed treatments pooled. LSD was used to determine the significance of pair wise differences between zones. This analysis design did not permit testing for an interaction between zone and seeding treatment.

Transformations were applied to the data as needed to improve the fit of the data to distributional assumptions of normality (ANOVA) or sphericity (MANOVA). Species richness data were $\log_{10}$ transformed in order to achieve equality of variances. Simpson's diversity data had equal variances and therefore were not transformed. The abundance of plant functional types was $\log_{10}(x+1)$ transformed in order to make it more spherical. The surface cover data were spherical except for bare soil, which was subsequently $\log_{10}(x+1)$ transformed.

2.4.2 Environmental Conditions

Four environmental variables were used to identify differences in terrain conditions: relative elevation, active layer depth, soil conductivity, and organic layer depth. These variables were assumed to be correlated with each other. For the lowland terrain, I used a split-plot MANOVA to test for the effects of both seeding treatment and zone on environmental variables. I transformed the relative elevation and salinity data using a $\log_{10}(x+1)$ transformation, which made all the variables spherical. The data were then normalized using Z-scores. For the upland terrain, I used the cap zone data to run a MANOVA to test if there was a significant difference between seeding treatments on the environmental variables. The data were all spherical and therefore not transformed. Following test results indicating no significant seeding treatment effect, I ran a MANOVA using the entire upland data set where the unseeded and seeded treatments were pooled.

The disturbance footprint of each sump site was estimated assuming that sumps were circular in shape. The length of the cap zone from the center of the sump to the edge of the perimeter (radius) was averaged for both transects on each sump (Table 2.1). I then calculated the disturbed area of the sump cap (km^2) using this averaged radius. In order to estimate the footprint of the sump including both the cap and perimeter zones, the radius was calculated using the average transect distance from the center of the sump cap to the edge of the undisturbed zone for each site.

Table 2.1: Sump sites in the lowland and upland terrain divided into unseeded and seeded sites. The latitude and longitude coordinates are included as well as the date the sump was decommissioned. The diameter of the sump was averaged from the two transects for each site.

Site Treatments		Site ID	Latitude	Longitude	Release Date	Cap diameter (m)	Sump Perimeter diameter (m)	Cap and perimeter diameter (m)
Lowland	Unseeded	C-42	69.3514	-134.9472	Nov-72	89	89	178
Lowland	Unseeded	E-58	69.2915	-135.2487	Jun-77	59	106	165
Lowland	Unseeded	D-43	69.3705	-134.9501	Sep-73	49	49	98
Lowland	Unseeded	H-54	69.3889	-134.9683	Apr-77	71	93	164
Lowland	Unseeded	B-19	69.3031	-135.3053	Feb-76	77	97	174
Lowland	Seeded	K-26	69.0917	-135.1042	Feb-73	47	91	138
Lowland	Seeded	I-22	69.1937	-135.3409	Mar-73	63	107	170
Lowland	Seeded	O-54	69.2326	-134.9753	Apr-74	10	64	74
Lowland	Seeded	C-58	69.2850	-135.2317	Oct-73	51	93	144
Upland	Unseeded	F-09	68.9745	-133.5293	Apr-72	62	81	143
Upland	Unseeded	K-16	69.2591	-135.0662	Jul-75	68	92	160
Upland	Seeded	J-06	69.2600	-135.0161	May-74	97	79	176
Upland	Seeded	D-58	68.9537	-133.4976	Mar-75	67	71	138

Table 2.2: ANOVA table for the split-plot design used to test the effect of seeding and zone in the lowland sites. Seeding treatment (A) is a fixed factor (p = 2 levels) with sites (B; q = 4/treatment) nested within each treatment. Zone (C) are groups of different levels of disturbance (r = 3; cap, perimeter, and undisturbed) that were selected without bias during oil and gas exploration.

Between sites	**df**	**Calculated df**
Seed treatment (A)	p-1	2-1 = 1
Sites within seed treatment (B(A))	p(q-1)	2(4-1) = 6
Within sites	**df**	**Calculated df**
Zones (C)	r-1	3-1 = 2
Zones x seeding (A x C)	(p-1) (r-1)	(2-1) (4-1) = 2
Sites within seeding x zone (B(A) x C)	p(q-1) (r-1)	2(4-1) (3-1) = 12
Residual	pqr (n-1)	2*4*3(0-1) = 0

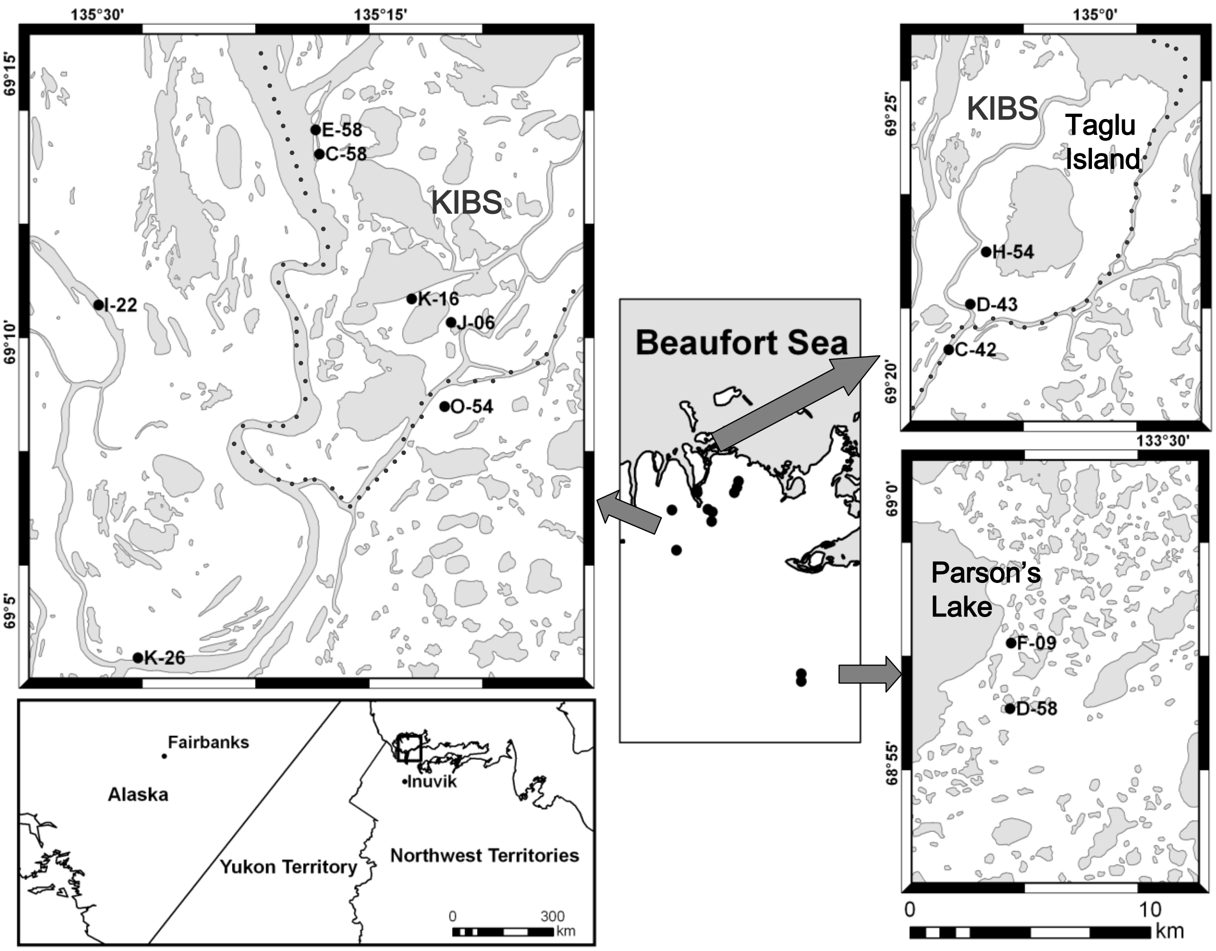


Figure 2.1: Map of the Mackenzie River Delta area, Northwest Territories. Sump study sites were located near Parson's Lake and in the Kendall Island Migratory Bird Sanctuary (KIBS, represented by the dotted line) on Taglu and surrounding islands. Black circles indicate site locations. Water bodies (lakes, river channels, and ocean) are gray and land is in white. Map produced in ArcGIS by Eric Vander Wal.

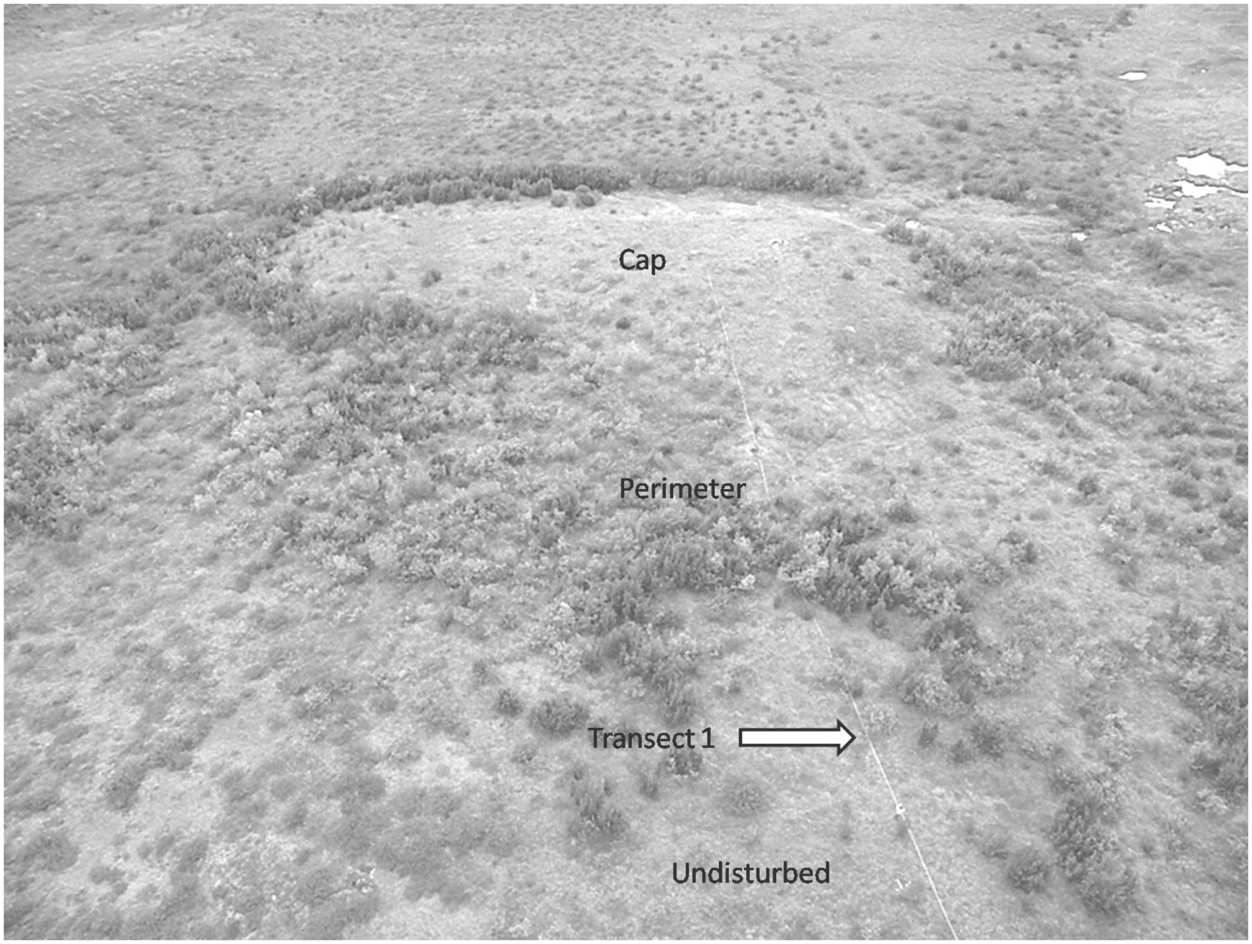


Figure 2.2: Aerial photo of upland site F-09 with one transect. Each zone of the sump (cap, perimeter, and undisturbed) is labeled. Photo credit: Julian Kanigan, 24 July 2008.

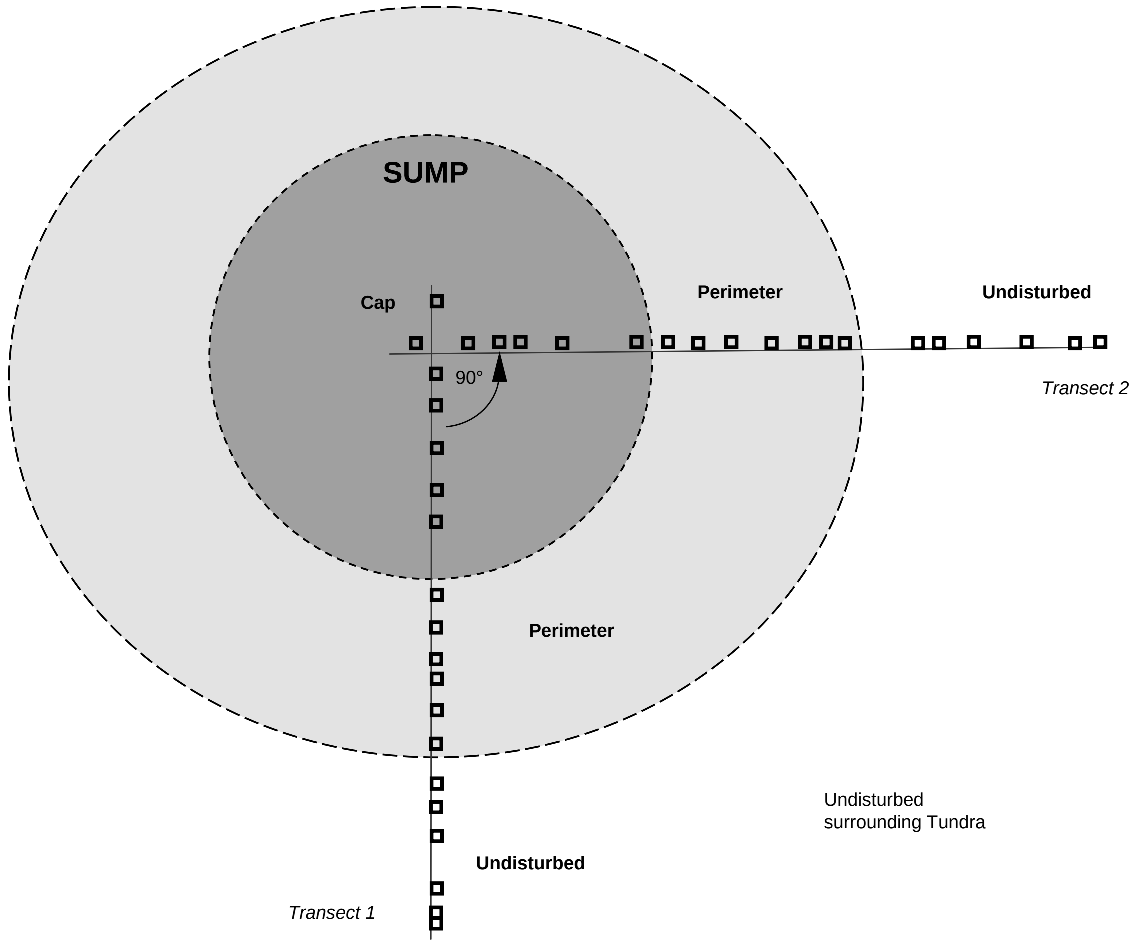


Figure 2.3: Aerial view of a sump with three zones: cap, perimeter, and undisturbed. The cap and perimeter make up the sump and the undisturbed zone is in the surrounding undisturbed tundra. Each sump has two transects divided into the three zones with six quadrats (open squares) randomly placed per zone for vegetation sampling. Note that the drawing is not to scale.

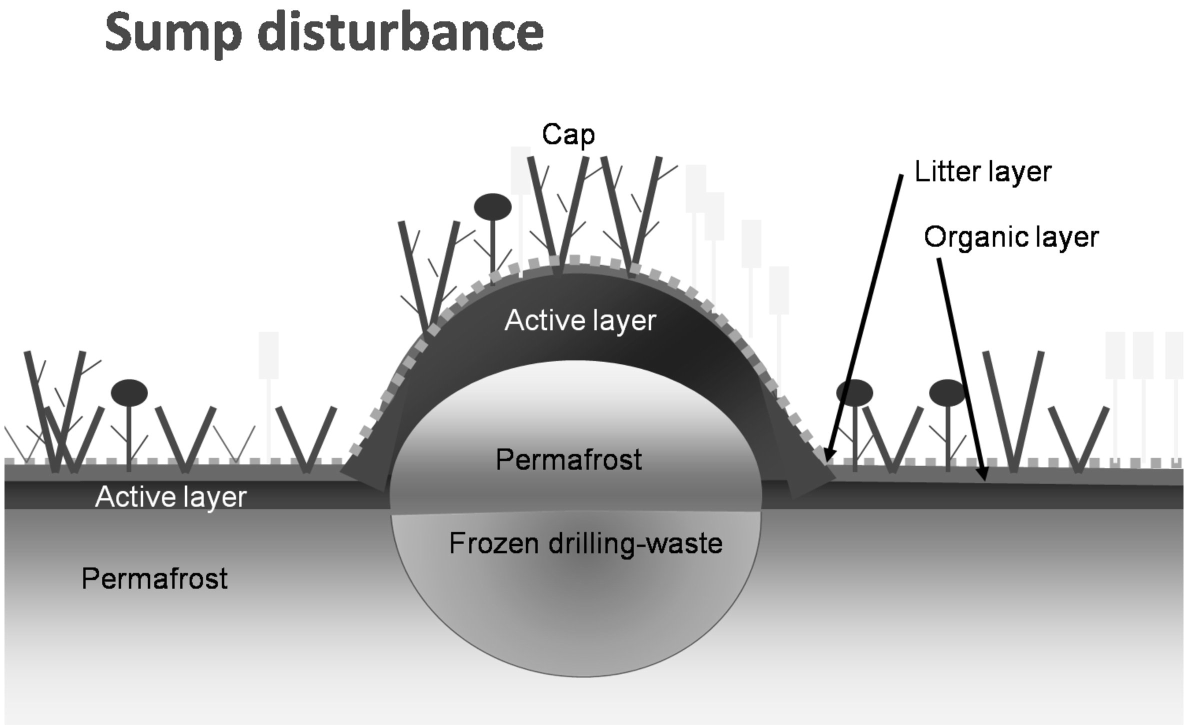


Figure 2.4: Schematic diagram of a sump disturbance in longitudinal section with the layers of ground illustrated for the elevated sump cap. Sumps are designed to keep the drilling-waste frozen in the permafrost. The seasonal thaw layer (active layer) of soil is shown in brown and above it is the organic soil layer followed by the litter layer and live vegetation.

3. Results

3.1 Vegetation Patterns

There was wide variation in the shape, size, and state of the sump (intact, slumping, or collapsed) across the sites. Generally the sumps were partially intact, but most had some amount of ponded water around the sump. Sumps were generally circular or elliptic in shape. They ranged in size from approximately 10 m (collapsed sump) to almost 100 m in diameter, but averaged 61 m in diameter (Table 3.1).

The approximate disturbance footprints of the sumps (Table 3.1) were calculated for each site from the measured extent of the cap and perimeter zones. The average area within the sump cap zone was 3,177 $\pm$ 555 m^2. Some smaller caps, such as site O-54 (lowland, seeded) had almost entirely collapsed into a surrounding pond. The sump caps in KIBS from this study include 0.0299 km^2 of the total land area (335 km^2) in KIBS. The footprint of a sump including the both cap and perimeter (assuming the area is circular) was considerably larger than the estimate that included only the cap, with an average area of 17,877 $\pm$ 1773 m^2. In KIBS, the total area within sump caps and perimeters from the 12 sites in this study was estimated to be 0.16 km^2.

Quadrat sampling at the sump sites identified a total of 94 species representing 56 genera in 25 plant families (Table 3.2). The most species-rich family was the Poaceae with 10 genera and 21 species. The Salicaceae also had a high diversity and were represented by 13 different *Salix* species. None of the species identified in quadrat sampling were listed by the Committee On the Status of Endangered Wildlife In Canada (COSEWIC), the Species at Risk Act (SARA) NWT, or SARA federal data bases (Government of Canada 2009). Most of the species were identified as either “secure” or “not assessed” on the NWT Species Infobase (GNWT 2009). *Salix hastata* L. (Halberd Willow) was listed as “sensitive” because it has a restricted distribution in the NWT (GNWT 2009). *Festuca lenensis* Drowbow (Tundra Fescue), and *Poa pseudoabbreviata* Rosh. (Shortcoat Bluegrass) were listed as “may be at risk” due to their very restricted distribution in the NWT (GNWT 2009). *F. rubra* L. (Red Fescue), *F. trachyphylla* (Hack.) Krajina (Hard Fescue), and *Alopecurus pratensis* L. (Meadow Foxtail) were all listed as “exotic/alien” and have been found along pipelines, road sides and in the upper Mackenzie River (GNWT 2009).

Two of the grasses seeded by Younkin and Martens (1976), *Festuca rubra* and *Poa pratensis*, were found in both seeded and unseeded sites. These grass species had been previously introduced into the NWT (Porsid and Cody 1980). *F. rubra* was present on the sump caps and/or perimeters of three seeded sites in the lowland terrain: K-26 (perimeter = 2/12 quadrats), O-54 (cap = 3/12, perimeter = 3/12), and C-58 (cap = 1/12, perimeter = 3/12). *F. rubra* was also found in the cap of the unseeded lowland site C-42 (cap = 1/12). At the upland sites, *F. rubra* was present on the sump caps of the seeded site D-58 (cap = 5/12) and unseeded site F-09 (cap = 1/12). *P. pratensis* was found at the seeded lowland site C-58 (perimeter = 1/12) as well as the unseeded site C-42 (perimeter = 4/12). In the upland, *P. pratensis* was present on the sump cap of the seeded site J-06 (cap = 2/12). No seeded or alien/exotic species were found in the undisturbed zone of the sumps.

Several of the sampled species only occurred in one disturbance zone across all of the sites (Table 3.3).There were nine species only found on the sump cap: three non-legume forb species, three grass species (including *Festuca lenensis* and *F. trachyphylla*), a sedge, a willow, and an evergreen shrub. The perimeter supported two species of non-legume forbs, a grass (*Poa* *psuedoabbreviata*), and a willow species. Six species were only found in the undisturbed zone, including one non-legume forb species, one grass, two sedges, and two willows (one of which was *Salix hastata).*

Ordination of the combined lowland and upland species frequency data resulted in a three dimensional NMS solution that captured ~86 % of the variation in the original distance matrix (Figure 3.1) The relative positioning of sample sites and zones in the ordination provides information on the relative compositional similarity between sites and disturbance zones within sites. The two terrain types occupied different areas of multivariate space in this ordination regardless of seeding treatment (Figure 3.1). The results of an MRPP test using terrain as the grouping variable indicated that the terrain types were significantly different ($A = 0.254$, $P < 0.001$). The lowland seeded sites occupied similar areas of ordination space, but the seeding treatment had no consistent pattern for both terrain types (Figure 3.1).

For some sites, zones within the site were most similar in composition to each other (i.e. close together in the ordination), such as lowland seeded sites I-22, K-26, C-58, and O-54 and upland unseeded sites F-09 and K-16, and unseeded D-58. In addition, sites I-22 and K-26 were similar to each other. Other sites had zones which occupied very different areas in the ordination space. The upland seeded site J-06 and the lowland unseeded sites H-54, C-42, E-58, and D-43 had caps that were distant from the perimeter or undisturbed zones in the ordination plot. This indicates that the vegetation composition at these sites differed substantially among disturbance zones. Generally, the perimeter and undisturbed zones of a site were closely associated. Upland site J-06 had a cap which was more closely related to the undisturbed zone, and the perimeter zone was plotted near other lowland zones (Figure 3.1).

Non-parametric correlations of environmental and surface cover variables with ordination axes revealed several factors that were related to variations in plant community composition, including relative elevation, lichen, water, bare soil, and moss cover, canopy height, active layer depth, conductivity, and organic layer depth (Table 3.4). Lowland sites were generally associated with greater active layer depth, conductivity, canopy height, water cover, and bare soil cover. Upland sites were associated with lower canopy height, higher relative elevation, shallower organic layer depth, and low cover of lichen and moss. However, the upland terrain was only represented by four sites and therefore the comparison between lowland and upland terrain should be considered preliminary.

Indicator species analysis for terrain types (lowland and upland) revealed eight significant indicators of lowland sites and 22 significant indicators of upland sites (Table 3.5). The lowland indicators include species from five families representing one legume, one non-legume forb, two grasses, two sedges/rushes, and two deciduous shrub species. Several of these species are broadly typical of lowland wet tundra including *Carex aquatilis* Wahlenb. (Water Sedge) and *Eriophorum angustifolium* Honckeny (Tall Cottongrass (Bliss 1988). The upland indicators come from 15 families comprised of 10 non-legume forb species, one grass, one sedge/rush, four deciduous shrubs, and six evergreen shrub species. Many of these species are typical of upland shrub-heath tundra as well as other widely distributed species including *Poa arctica* R. Br. (Arctic Bluegrass), *Alnus viridis* subsp. *crispa* (Ait.) Tirrill (Mountain Alder) and *Salix glauca* L. (Grayleaf Willow; Hernandez 1973, Forbes et al. 2001).

3.1.1 Lowland Terrain

Ordination of lowland sites alone (n = 8 sites x 3 zones) indicated that the species community composition differed between the cap and undisturbed zones, with perimeter communities intermediate between the two (Figure 3.2). Three seeded sites had caps that occupied quite distinct areas of the ordination space compared to the four unseeded caps, but the trend was not consistent. There were other zones (perimeter and undisturbed) grouped with these caps. For example, I-22 and K-26 had all three zones clustered with the seeded caps. The disturbance regime was similar at these two sites because they were both eroding from frequent flooding. A hierarchical cluster analysis also revealed that three of the caps of the seeded sites clustered together, but C-58 did not (Figure 3.3). This suggests that seeded caps had similar vegetation composition. At a broader scale, all of the caps (both unseeded and seeded) clustered together (20% information remaining) except for the cap of seeded site C-58 (Figure 3.3). Site C-58 had a collapsed sump cap with almost no relief (Table 3.1). Site C-58 had all three zones clustered closed to each other, as did site I-22, which indicates that the zones at these sites were more similar to each other than to similar zones at other sites. Site H-54 had a cap zone that was distinct from all other sites. The undisturbed zone in K-26 and I-22 did not cluster with the other undisturbed zones suggesting that their composition was not the same as most other undisturbed zones. However, in general, the disturbance responses were often site specific. Sites such as I-22 and K-26 tended to be clustered together probably because these sites had evidence of frequent flooding such as a low organic layer and the presence of driftwood at the sites. C-58 also had evidence of frequent flooding, which indicates that natural disturbance at this site may account for the vegetation composition difference from the rest of the seeded caps. The undisturbed zone for the unseeded sites, along with undisturbed zone for seeded sites O-54 and C-58 clustered together.

All of the environmental and surface cover variables except litter cover had strong correlations ($\tau > 0.200$) with at least one of the ordination axes (Table 3.6). Line vectors on the ordination illustration indicated that cap zones were associated with higher relative elevation, greater canopy height, and thicker active layers (Figure 3.2). Most undisturbed zones (except I-22 and K-26) were associated with a thicker organic layer, high total vegetation and lichen cover, low salinity, low bare soil cover, and a thin active layer.

The results of the indicator species analysis identified two indicator species for the seeded sites: *Equisetum arvense* L. (Field Horsetail; IV = 71.0, P = 0.006) and *Hedysarum alpinum* L. (Alpine Sweetvetch; IV = 65.2, P = 0.005). There were several indicator species for each disturbance zone (Table 3.7). These species are indicators of each zone because they were found at higher frequency and abundance in that zone, but may also occur in other zones. The cap zone had six significant indicator species, including one legume, three herb, one grass, and one deciduous shrub species. *Arctagrostis latifolia* (Wideleaf Polargrass) is a ubiquitous species in the tundra, and *Salix alaxensis* (Feltleaf Willow) and *Oxytropis deflexa* (Drooping Locoweed) are commonly found together on gravelly stream banks and lake shores (Porsid and Cody 1980). *Lomatogonium rotatum* (Marsh Felwort) can be found near saline lakes or springs, *Parnassia palustris* (Marsh Grass of Parnassus) is common in wet calcareous soils and *Pyrola grandiflora* (Largeflowered Wintergreen) is abundant on sunny tundra slopes (Porsid and Cody 1980). All of these species, except *O. deflexa,* which was only found on the sump caps in the lowland, also occur in the undisturbed tundra, but were less abundant there. The perimeter had no significant indicators and the undisturbed zone had only one indicator species, the sedge *Eriophorum angustifolium* (Tall Cottongrass).

The seeding treatment had no significant effect on species richness at the lowland sites (split-plot ANOVA, $F_{1, 6} = 1.911$; P = 0.216; Figure 3.4). Species richness was significantly different between the three zones (split-plot ANOVA, $F_{2, 12} = 8.407$; P = 0.005). The cap had significantly more species than both the perimeter and undisturbed zones ($P < 0.01$; Figure 3.4). Seeding treatment had no significant effect on Simpson's diversity (split-plot ANOVA, $F_{1, 6} = 5.151$; $P = 0.151$; Figure 3.5). However, there was a significant difference in Simpson's diversity between the zones (split-plot ANOVA, $F_{2, 12} = 7.019$; $P = 0.010$; Figure 3.5). The cap had significantly greater diversity than the undisturbed zone, but there was no difference between the perimeter and either the cap or undisturbed zone ($P < 0.002$).

Consistent with the analysis of total species richness, seeding treatments did not significantly affect the species richness within different plant functional types (split-plot MANOVA, $F_{5,2} = 0.294$; $P = 0.883$). The zones had a significant effect on the richness of different functional types (split-plot MANOVA, $F_{10, 18} = 2.602$; $P = 0.037$; Figure 3.6). There was a significant difference in richness of four functional types between zones (split-plot ANOVAs, Table 3.8). There was no difference in the richness of sedges/rushes across the zones. There were more legumes, non-legume forbs, grasses, and deciduous shrub species on the caps compared to the undisturbed zones.

There was no significant difference in surface cover between unseeded and seeded treatments (split-plot MANOVA, $F_{4,3} = 1.679$; $P = 0.350$), but zones differed (split-plot MANOVA, $F_{8, 20} = 10.718$; $P < 0.001$; Figure 3.7). This difference was largely due to decreased cover of total vegetation and moss (split-plot ANOVAs, Table 3.9). Unlike the richness and diversity measures, the total vegetation cover was significantly greater in the undisturbed zone compared to the cap. The moss cover was similar on the cap and undisturbed zones, but significantly less on the perimeter compared to the cap and undisturbed zones.

3.1.2 Upland Terrain

The NMS ordination for only upland sites (n = 4 sites x 3 zones) resulted in a two-dimensional solution that captured 88 % of the variation in the original distance matrix (Figure 3.8). A graph of the two axes illustrates that the cap and undisturbed zones occupied discrete areas of ordination space (species composition), whereas the perimeter zone was dispersed throughout the ordination. The unseeded and seeded sites did not occupy discrete areas of ordination space. Sites J-06 (seeded), D-58 (seeded), and F-09 (unseeded) have similar caps whereas K-16 has a cap composition that was more similar to the perimeter of J-06. The spatial proximity of sites was apparent in the ordination, as the undisturbed zone of F-09 (unseeded) and D-58 (seeded) were closely associated and were both located near Parson's Lake, whereas the outer delta sites K-16 (unseeded) and J-06 (seeded) were most similar to each other.

Several environmental variables had strong correlations with the ordination axes including conductivity and organic layer depth (Table 3.10). Undisturbed zones were associated with a thick organic layer and greater cover of total vegetation and lichen (Figure 3.8). High soil conductivity and high relative elevation were associated with cap zone. The cap zone of site K-16 was similar in composition to its perimeter zone and was associated with a higher cover of standing water and was not grouped as closely together with the other cap zones. Site D-58 (seeded) was interesting because it showed an almost straight line pattern from the cap associated with a deeper active layer and higher soil conductivity, to the perimeter and then the undisturbed zone with a shallow active layer, low soil conductivity, higher lichen cover, and deep organic layer.

The indicator species analysis for seeding treatment revealed only one indicator species: *Salix richardsonii* Hook. (Richardson's Willow; IV = 72.2, P = 0.05) for the unseeded sites and no indicator species for the seeded sites. Disturbance zones revealed that, like the lowland sites, only the cap and undisturbed zones had species with significantly higher abundances and frequencies that could be considered indicators of those zones (Table 3.11). The herb *Epilobium angustifolium* subsp. *angustifolium* (Fireweed) had a significant indicator value for the cap zone. *E. angustifolium* subsp. a*ngustifolium* is a pioneer species of disturbed areas in the NWT (Porsid and Cody 1980), but was found in all zones of the sump. There were five significant indicator species for the undisturbed zone including the non-legume forb *Arctostaphylos rubra* (Red Fruit Bearberry), and evergreen shrubs *Empetrum nigrum* (Black Crowberry) and *Ledum palustre* subsp. *decumbens* (Marsh Labrador Tea).

Species richness was not significantly different between seeding treatments (cap zone only ANOVA, $F_{1,2} = 0.472$; $P = 0.563$), but was significantly different between disturbance zones (ANOVA, $F_{2,9} = 5.120$; $P = 0.033$). Contrary to the lowland sites, the undisturbed zone ($\bar{x} = 24.75 \pm 3.82$) had greater richness than both the cap ($\bar{x} = 18.25 \pm 1.25$; $P = 0.039$) and perimeter ($\bar{x} = 19.00 \pm 4.04$; $P = 0.014$). Simpson's diversity was not significantly different between seeding treatments (cap zone only ANOVA, $F_{1,2} = 2.617$; $P = 0.247$), or between the zones (ANOVA, $F_{2,9} = 1.277$; $P = 0.325$).

There was also no difference in the richness of plant functional types between seeding treatments (cap zone only MANOVA, $F_{1,2} = 0.500$; $P = 0.707$) and between zones (MANOVA, $F_{6,4} = 0.925$; $P = 0.558$).

Seeding treatment had no effect on surface cover (cap zone only MANOVA, $F_{1,2} = 2.071$; $P = 0.441$). Disturbance zones also did not show significant differences in four surface cover variables: total vegetation, litter, bare soil, and moss cover (MANOVA, $F_{8,14} = 1.166$; $P = 0.383$).

3.2 Environmental Conditions

There was no difference in environmental variables (elevation, active layer depth, soil conductivity, and organic layer depth) between seeded and unseeded treatments for lowland (split-plot MANOVA, $F_{4,3} = 2.298$; $P = 0.260$) or upland terrain (cap zone only MANOVA, $F_{1, 2} = 8.631$; $P = 0.234$). However, there was a significant difference in environmental variables between zones for the lowland sites (split-plot MANOVA, $F_{8, 20} = 17.337$; $P < 0.001$; Figure 3.9). There was a marginally significant difference between zones of the upland sites (MANOVA, $F_{8, 14} = 2.536$; $P = 0.061$).

Relative elevation was significantly different between zones at the lowland sites (split-plot ANOVA, $F_{2, 12} = 1405.76$; $P < 0.0001$; Figures 3.9, 3.10) with the highest elevations recorded in the cap followed by the perimeter and then the undisturbed zones. This effect was the same in the upland sites (ANOVA; $F_{2, 9} = 13.151$; $P = 0.002$; Figure 3.10), where the cap had a significantly higher relative elevation than both the perimeter ($P = 0.009$) and undisturbed zones ($P = 0.001$). The perimeter zone in most sites had some standing water or large pilings where temporary buildings once stood. Some of the caps were actively eroding into the standing water in the perimeter (Lowland sites: D-43, I-22, and O-54; Upland site F-09; Figure 3.11).

The active layer followed a similar pattern to elevation across zones in both terrain types (Figure 3.12). In the lowland sites, active layer depth varied significantly among zones (split-plot ANOVA; $F_{2, 12} = 8.321$; $P = 0.005$), and was 89 and 56 % thicker in the cap and perimeter, respectively, than in the undisturbed zone (Figure 3.9). The active layer was also significantly different between zones in the upland (ANOVA, $F_{2, 9} = 5.877$; $P = 0.023$) with the cap significantly deeper than the undisturbed zone ($P = 0.008$), but no difference between the perimeter and undisturbed zones (Figure 3.12).

Soil conductivity in the lowland sites was high primarily due to three sites (E-58, K-26, and C-58) that were severely saline (Figure 3.13). Also, the cap and perimeter of several sites, including the lowland sites D-43 and O-54 and upland D-58, had saline soil patches, which had a crust of salt on the surface with no vegetation. The mean soil conductivity for the sump cap, perimeter, and undisturbed were significantly different (split-plot ANOVA, $F_{2, 12} = 944.044$; $P < 0.001$; Figure 3.9). The undisturbed zone was the only zone that consistently had a mean salinity within the range of non-saline soils (0-2 dS/m) (Dunn 2001). The perimeter had the highest soil conductivity and was significantly different from both the cap and undisturbed zone. The results were similar in the upland terrain with a significant difference in soil conductivity within the zones (ANOVA, $F_{2, 9} = 4.263$; $P = 0.050$; Figure 3.13). The cap and perimeter zones were more saline than undisturbed ($P = 0.031$ and $P = 0.034$, respectively), but there was no difference in conductivity between the cap and perimeter zones at upland sites.

The organic layer depth for both terrain types varied (Figure 3.14). Several lowland sites had organic layer depths that were not consistent between the zones because of either frequent flooding in the area, or very wet marshy conditions in which we could not take a soil core sample (e.g. standing water). Sites H-54 and I-22 had no data for the perimeter and undisturbed zones due to the extremely wet marsh conditions. Nevertheless the organic layer depth between zones was marginally significantly different for the lowland sites (split-plot ANOVA, $F_{2, 12} = 318.10$; $P = 0.057$; Figure 3.9). The cap had a significantly smaller organic layer compared to both the perimeter and undisturbed zone, but there was no significant difference between the cap and perimeter. The results were similar for the upland sites, which had significantly different organic layers between the zones (ANOVA, $F_{2, 9}= 9.562$; $P = 0.006$; Figure 3.14) with the cap (P =

0.002) and perimeter ($P = 0.011$) having a poorly developed organic layer compared to the undisturbed zone.

At most sites, there was a relatively clear visual distinction between disturbed zones (cap, perimeter) and surrounding undisturbed tundra. The vegetation community data was inspected for evidence of obvious edge effects of disturbance that may have permeated into the undisturbed zone. At some sites, the active layer appeared to be slightly deeper in the first two quadrats sampled in the undisturbed zone (2-10 m) than subsequent measurements. Some sites also showed a higher cover of standing water at sampling locations in the undisturbed zone that were closer to the sump perimeter. This effect was not consistent with each site, but was observed for sites in both the lowland and upland terrain. Thus, although physical disturbance effects may have been adequately separated by the delineation of disturbance zones, edge effects of disturbance on the soil thermal regime or drainage may actually extend up to 10 m beyond the sump perimeter.

Table 3.1: List of sump sites with the site name (e.g. C-42). Sump open in the summer refers to if the sump and well-site had summer drilling activities. Ponding refers to the presence of standing water in the perimeter associated with the sump. Notes on the site can include the sump condition, for example collapsing refers to the slumping and degradation of the sump. The approximate disturbance footprints (m^2) are based on the area of the sump assuming it is circular in shape.

		Site Name	Sump open in Summer	Ponding	Notes on the site	Disturbance footprint of sump cap (m^2)	Disturbance footprint of sump cap and perimeter (m^2)
Lowland	Unseeded	C-42	yes	moderate	Collapsing sump	6,221	24,884
Lowland	Unseeded	E-58	no	moderate	Sump cap intact	2,734	21,382
Lowland	Unseeded	D-43	yes	severe	Collapsing sump	1,885	7,542
Lowland	Unseeded	H-54	no	moderate	Sump cap intact	1,260	21,124
Lowland	Seeded	K-26	no	severe	Sump cap intact, site frequently flooded.	1,735	14,957
Lowland	Seeded	I-22	no	severe	Collapsed sump, site actively eroding into river channel.	3,117	22,698
Lowland	Seeded	O-54	no	severe	Collapsed sump, sump very small.	79	4,301
Lowland	Seeded	C-58	yes	severe	Collapsing sump, almost no relief.	2,043	16,286
Upland	Unseeded	F-09	no	moderate	Collapsing sump	3,019	16,060
Upland	Unseeded	K-16	yes	minor	Sump cap intact	3,632	20,106
Upland	Seeded	J-06	no	severe	Collapsed sump	7,389	24,328
Upland	Seeded	D-58	no	none	Sump cap intact	3,526	14,957

Table 3.2: List of plant species encountered in quadrat samples of Lowland and Upland sites organized by taxonomic order and family. The X represents species presence in that site. Species are considered secure or not assessed by the GNWT unless otherwise listed (GNWT 2009). Codes in brackets next to each family name indicate membership within the following plant functional groups: L = legumes, F = forbs, G = grasses, S/R = sedges/rushes, DS = deciduous shrubs, and ES = evergreen shrubs.

	Lowland									Upland			
	Unseeded					Seeded				Unseeded		Seeded	
	C42	D43	E58	H54	B19	C58	O54	K26	I22	K16	F09	J06	D58
Equisetopsida													
Equisetaceae [F]													
Equisetum arvense L.	X	X	X	X	X	X	X	X	X	X	X	X	X
E. palustre L.									X				
E. variegatum Schleich. ex F. Weber & D.M.H. Mohr	X	X	X	X		X			X	X		X	
Dicotyledons													
Asteraceae [F]													
Artemisia tilesii Ledeb	X							X	X	X		X	
Aster sibiricus. L.								X					
Tripleurospermum maritima (L.) W.D.J. Koch subsp. *phaeocephala* (Rupr.) Hamet-Ahti		X											
Petasites frigidus (L.) Fries											X		X
P. sagittatus (Banks ex Pursh) Gray										X			
Saussurea angustifolia (Willd.) DC.				X		X				X	X	X	X
Senecio lugens Richards.												X	
Tephroseris atropurpurea (Ledeb.) Holub												X	
Betulaceae [DS]													
Alnus viridis (Vill.) Lam. & DC. subsp. *crispa*(Ait.) Turrill							X			X	X		X
Betula nana L.			X				X			X	X	X	X
Brassicaceae [F]													
Cardamine digitata Richards.	X											X	
C. pratensis L.	X												
Caryophyllaceae [F]													
Stellaria longipes Goldie						X	X			X	X	X	X
Empetraceae [ES]													
Empetrum nigrum L.										X	X	X	
Ericaceae [ES-DS]													
Arctostaphylos rubra (Rehd. & Wilson) Fern.			X					X		X	X	X	X
Cassiope tetragona (L.) D. Don												X	
Chamedaphne calyculata (L.) Moench										X		X	
Ledum palustre L. subsp. *decumbens* (Ait.) Hultén										X	X	X	
Vaccinium uliginosum L.	X										X	X	
V. vitis-idaea L.								X		X	X	X	
Rhododendron lapponicum (L.) Wahlenb.												X	

	Lowland									Upland			
	Unseeded					Seeded				Unseeded		Seeded	
Dicotyledons	C42	D43	E58	H54	B19	C58	O54	K26	I22	K16	F09	J06	D58
Fabaceae [L]													
Astragalus alpinus L.		X	X	X		X					X		
Hedysarum alpinum L.		X				X	X	X	X				
Lupinus arcticus S. Wats.										X		X	
Oxytropis deflexa (Pallas) DC.	X	X	X	X			X			X	X	X	X
Gentianaceae [F]													
Lomatogonium rotatum (L.) Fries ex Fern.	X	X	X		X	X	X						
Hippuridaceae [F]													
Hippuris vulgaris L.	X		X		X								
Lentibulariaceae [F]													
Pinguicula vulgaris L.				X						X			
Onagraceae [F]													
Epilobium angustifolium L. subsp. *angustifolium*				X		X				X	X	X	X
Polygonaceae [F]													
Polygonum viviparum L.		X		X		X	X		X	X	X	X	X
Pyrolaceae [F]													
Pyrola grandiflora Radius	X	X	X				X	X	X	X	X	X	
Orthilia secunda (L.) House											X	X	
Ranunculaceae [F]													
Caltha palustris L.	X	X				X	X						
Ranunculus cymbalaria Pursh					X								
Rosaceae [ES-F]													
Dryas integrifolia Vahl.										X		X	
Rubus chamaemorus L.						X				X		X	X
Salicaceae [DS]													
Salix alaxensis (Anderss.) Coville	X		X	X	X	X	X	X	X	X		X	
S. arbusculoides Anderss.		X								X	X	X	
S. arctophila Cockerell ex Heller				X		X							
S. bebbiana Sarg.	X												
S. boothii Dorn						X							
S. fuscescens Anderss.													X
S. glauca L. *	X	X	X					X		X	X	X	X
S. hastata L. **(sensitive)**	X												
S. niphoclada Rydb.	X	X			X	X			X				
S. planifolia Pursh				X					X			X	X
S. pulchra Cham.	X				X					X			X
S. reticulata Hook.													X

	Lowland									Upland			
	Unseeded					Seeded				Unseeded		Seeded	
Dicotyledons	C42	D43	E58	H54	B19	C58	O54	K26	I22	K16	F09	J06	D58
Salicaceae [DS]													
S. richardsonii L.	X	X	X	X		X	X	X	X	X	X	X	
Saxifragaceae [F]													
Parnassia palustris L.	X	X			X	X	X	X	X	X		X	
Saxifraga hirculus L.											X		X
Schrophulariaceae [F]													
Castilleja elegans Malte		X	X	X			X			X			
C. raupii Pennell	X												
Pedicularis capitata M.F. Adams												X	
P. lanata Cham. & Schlecht. subsp. *lanata*											X	X	
P. langsdorfii Fisch. ex Stev.		X	X	X	X	X	X			X		X	
P. verticillata L.						X							
Monocotyledons													
Cyperaceae [S/R]													
Carex aquatilis Wahlenb.	X	X	X	X	X	X	X	X	X	X	X	X	
C. bigelowii Torr. ex Schwein.	X			X						X	X	X	X
C. capillaris L.	X	X	X	X		X		X	X				
C. lugens Holm										X			
C. microchaeta Holm **										X			X
Eriophorum angustifolium Honckeny	X	X	X	X	X	X	X		X	X			
E. vaginatum L.										X			
Kobresia myosuroides (Vill.) Fiori												X	
Juncaceae [S/R]													
Juncus arcticus Willd.						X				X			
J. balticus var. *littoralis* Engelm.					X								
Juncaginaceae [F]													
Triglochin palustre L.												X	
Liliaceae [F]													
Tofieldia pusilla (Michx.) Pers.		X		X						X		X	
Orchidaceae [F]													
Platanthera obtusata (Banks ex Pursh) Lindl.								X		X			
Poaceae [G]													
Alopecurus pratensis L. **(exotic/alien)**					X								
Arctagrostis latifolia (R. Br.) Griseb.	X	X	X	X	X	X	X	X	X	X	X	X	X
Arctophila fulva (Trin.) Rupr. ex Anderss.					X	X							
Calamagrostis lapponica (Wahlenb.) Hartman	X		X		X					X			
C. stricta (Timm) Koel.***		X	X	X	X	X	X	X	X			X	

	Lowland									Upland			
	Unseeded					Seeded				Unseeded		Seeded	
Monocotyledons	C42	D43	E58	H54	B19	C58	O54	K26	I22	K16	F09	J06	D58
Poaceae [G]													
Deschampsia caespitosa (L.) Beauv.					X				X				
Dupontia fisheri R. Br.		X											
Festuca lenensis Drobow **(may be at risk)**		X											
F. richardsonii Hook.	X	X	X	X	X	X	X	X	X	X		X	
F. rubra L. **(introduced/seeded)**	X					X	X	X			X		X
F. trachyphylla (Hack.) Krajina **(exotic/alien)**								X					
Poa arctica R. Br.										X	X	X	X
P. glauca Vahl.			X			X	X			X	X		
P. palustris L.			X	X	X								
P. pratensis L. **(introduced/seeded)**	X					X						X	
P. pseudoabbreviata Rosh. **(may be at risk)**								X					
Puccinellia agrostidea Sorensen		X			X								
P. angustata (R. Br.) Rand & Redf.													
P. arctica (Hook.) Fern. & Weatherby		X	X		X							X	
P. nutkaensis (J. Presl) Fern. & Weatherby					X								
Trisetum spicatum (L.) Richter										X			

**S. glauca* also identified as *S. glauca* var. *acutifolia* (Hook.) Schneid.
**also includes *C. microchaeta* Holm subsp. *nesophila* (Holm) E. Murr.
***also includes *C. stricta* Lam. subsp. *stricta* (Timm) Koel. and *C. stricta* Lam. subsp. *inexpansa* (Gray) C.W. Green

Table 3.3: List of species present in quadrat sampling from both lowland and upland terrain that were only found in one disturbance zone (cap, perimeter, and undisturbed). Species status was "secure" unless indicated by * according to NWT Species Infobase (GNWT 2009).

Cap	**Perimeter**	**Undisturbed**
Carex lugens	*Cardamine pratensis*	*Dupontia fisheri*
Castelleja elegans	*Equisetum palustre*	*Eriophorum vaginatum*
*Festuca leneasis**	*Poa pseudoabbreviata**	*Kobresia myosuroides*
*F. trachyphylla****	*Salix bebbiana*	*Petasites frigidus*
Pedicularis verticilla		*Salix hastata***
Puccinellia agrostidea		*S. reticulata*
Rhododendron lapponicum		
Salix boothii		
Triglockin palustre		

* Species status of May Be at Risk
** Species status of Sensitive
***Species status of Exotic/Alien

Table 3.4: Non-parametric correlation coefficients (Kendall's tau) between vegetation ordination axes and environmental variables and surface cover for both lowland/upland NMS ordination. Stronger correlations (τ>0.200) are shown in bold font.

Environmental Variable	Axis 1	Axis 2	Axis 3
Relative elevation	**-0.540**	0.003	-0.032
Total vegetation cover	0.010	-0.079	0.175
Lichen cover	**-0.416**	**-0.344**	-0.115
Litter cover	-0.150	-0.025	-0.045
Water cover	**0.220**	-0.105	-0.002
Bare soil cover	-0.019	**0.458**	-0.099
Moss cover	0.098	**-0.204**	-0.179
Canopy height	**-0.292**	**0.257**	0.102
Active layer depth	-0.010	**0.521**	0.016
Conductivity	0.052	**0.554**	-0.065
Organic layer depth	0.168	**-0.544**	-0.174

Table 3.5: Results of an analysis to determine indicator species using terrain as a grouping variable for combined lowland and upland species proportion data from quadrat sampling and 4999 randomizations. The indicator value (IV) and p value (P) is shown for each significant indicator species.

	Species	IV	P
Lowland			
	Carex aquatilis	78.3	< 0.001
	Salix richardsonii	69.7	0.007
	Eriophorum agnustifolium	64.8	0.005
	Calamagrostis stricta	64.1	0.004
	Festuca richardsonii	61.0	0.009
	Salix alaxensis	55.3	0.031
	Hedysarum alpinum	41.7	0.029
	Lomatogonum rotatum	41.7	0.026
Upland			
	Carex bigelowii	88.1	< 0.001
	Betula nana	79.8	< 0.001
	Arctostaphylos rubra	77.1	< 0.001
	Salix glauca	69.2	0.001
	Empetrum nigrum	63.6	< 0.001
	Ledum palustre subsp. *decumbens*	63.6	< 0.001
	Alnus viridis subsp. *crispa*	62.6	< 0.001
	Epilobium angustifolium	62.2	0.001
	Oxytropis deflexa	60.3	0.002
	Stellaria longipes	56.7	0.001
	Poa arctica	54.5	< 0.001
	Vaccinium uliginosum	53.5	0.001
	Vaccinium vitis-ideae	53.4	< 0.001
	Saussurea angustifolium	50.0	0.002
	Petasites frigidus	45.5	0.001
	Dryas integrifolia	36.4	0.007
	Orthilia secunda	36.4	0.006
	Salix arbusculoides	34.9	0.011
	Tofieldia pusilla	31.5	0.027
	Cassiope tetragona	27.3	0.027
	Pedicularis lanata subsp. *lanata*	27.3	0.023
	Saxifraga hirculus	27.3	0.024

Table 3.6: Non-parametric correlation coefficients (Kendall's tau) between vegetation ordination axes and environmental variables and surface cover for the lowland NMS ordination. Stronger correlations (τ>0.200) are shown in bold font.

Environmental Variable	Axis 1	Axis 2	Axis 3
Relative elevation	**-0.232**	**-0.529**	-0.022
Total vegetation cover	**-0.225**	-0.159	0.014
Lichen cover	**0.260**	-0.027	**-0.206**
Litter cover	-0.087	0.065	0.051
Water cover	**0.286**	**0.212**	-0.092
Bare soil cover	**-0.287**	**-0.387**	-0.080
Moss cover	**0.410**	-0.015	0.161
Canopy height	**-0.507**	**-0.384**	0.036
Active layer depth	**-0.464**	**-0.529**	-0.123
Conductivity	**-0.309**	**-0.367**	-0.011
Organic layer depth	**0.567**	**-0.385**	0.095

Table 3.7: Results of an analysis to determine indicator species using 4999 randomizations on the lowland species (proportion data) from quadrat sampling. The groups are the zones of the sump (cap, perimeter, and undisturbed). The indicator value (IV) and p value (P) is shown for each significant indicator species.

	Species	**IV**	**P**
Cap			
	Arctagrostis latifolia	71.4	< 0.001
	Oxytropis deflexa	62.5	0.006
	Salix alaxensis	61.9	0.004
	Pyrola grandiflora	57.1	0.017
	Lomatagonium rotatum	56.8	0.017
	Parnassia palustre	51.9	0.031
Perimeter			
	none		
Undisturbed			
	Eriophorum angustifolium	46.5	0.049

Table 3.8: Lowland split-plot ANOVA results for functional types based on differences between the zones of the sump (cap, perimeter, and undisturbed). Significant p-values are in bold. The star (*) indicates a marginally significant p-value.

	F	**df**	**P**
Legumes	8.776	2, 12	**0.004**
Non-legume forbs	4.261	2, 12	**0.040**
Grasses	5.628	2, 12	**0.019**
Deciduous shrubs	1.219	2, 12	**0.109***
Sedges/Rushes	2.677	2, 12	0.330

Table 3.9: Lowland split-plot ANOVA results for surface cover based on differences within the zones of the sump (cap, perimeter, and undisturbed). Significant p-values are in bold.

	F	**df**	**P**
Total vegetation cover	6.620	2, 12	**0.012**
Bare soil cover	2.204	2, 12	0.153
Moss cover	6.161	2, 12	**<0.001**
Litter cover	0.168	2, 12	0.847

Table 3.10: Non-parametric correlation coefficients (Kendall's tau) between vegetation ordination axes and environmental variables and surface cover for the upland NMS ordination. Stronger correlations ($\tau \geq 0.200$) are shown in bold font.

Environmental Variable	Axis 1	Axis 2
Relative elevation	**-0.303**	-0.182
Total vegetation cover	**0.394**	**0.273**
Lichen cover	**0.315**	**-0.448**
Litter cover	0.062	**-0.246**
Water cover	**-0.319**	**0.280**
Bare soil cover	**-0.349**	**-0.349**
Moss cover	0.168	**0.260**
Canopy height	-0.152	-0.030
Active layer depth	**-0.333**	-0.030
Conductivity	**-0.759**	-0.091
Organic layer depth	**0.758**	0.030

Table 3.11: Results from an analysis to determine indicator species using 4999 randomizations on the upland species (proportion data) from quadrat sampling. The groups are the zones of the sump (cap, perimeter, and undisturbed). The indicator value (IV) and p value (P) is shown for each significant indicator species.

Species	IV	P
Cap		
Epilobium angustifolium	73.7	0.017
Perimeter		
none		
Undisturbed		
Polygonum viviparum	92.9	0.009
Ledum palsutre subsp. *decumbens*	84.2	0.016
Arctostaphylos rubra	79.4	0.004
Rubus chaemaemorus	75	0.054
Empetrum nigrum	67.6	0.035

Figure 3.1: (following page): Distribution of both lowland and upland sites with species frequency data of a three dimensional NMS ordination. The top ordination is plotted on axis 1 and 2, and the bottom left ordination is plotted on axis 3 and 2. The NMS ordination used 200 iterations to produce a solution with a final stress of 9.73 and instability of 0.0004. The symbols indicate whether plots were located in lowland (squares) or upland (triangles) terrain, and the unfilled symbols indicate unseeded sites whereas the filled symbols indicate seeded sites. The lines in the top ordination connect the cap to the perimeter and the perimeter to the undisturbed zone within a site.

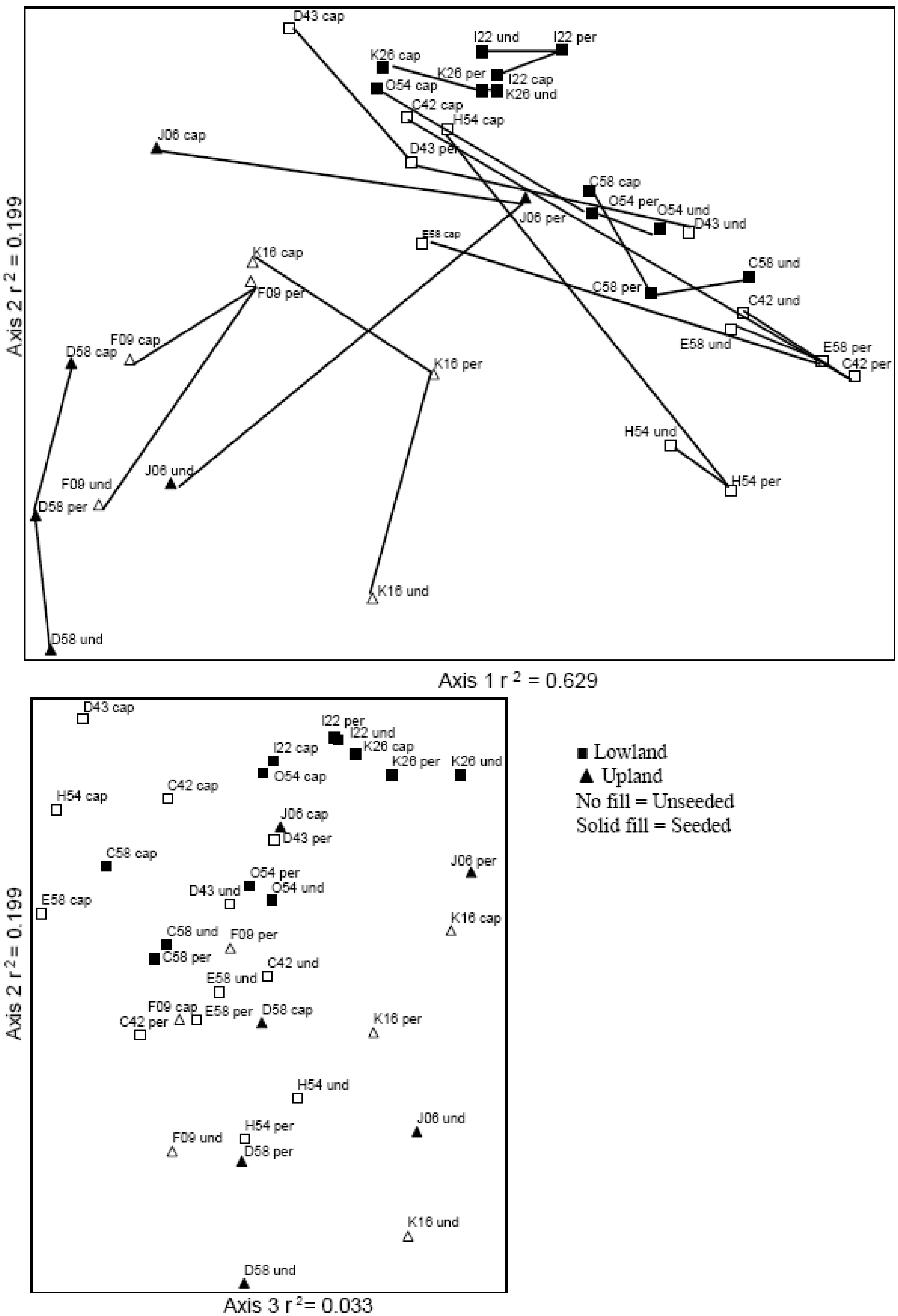
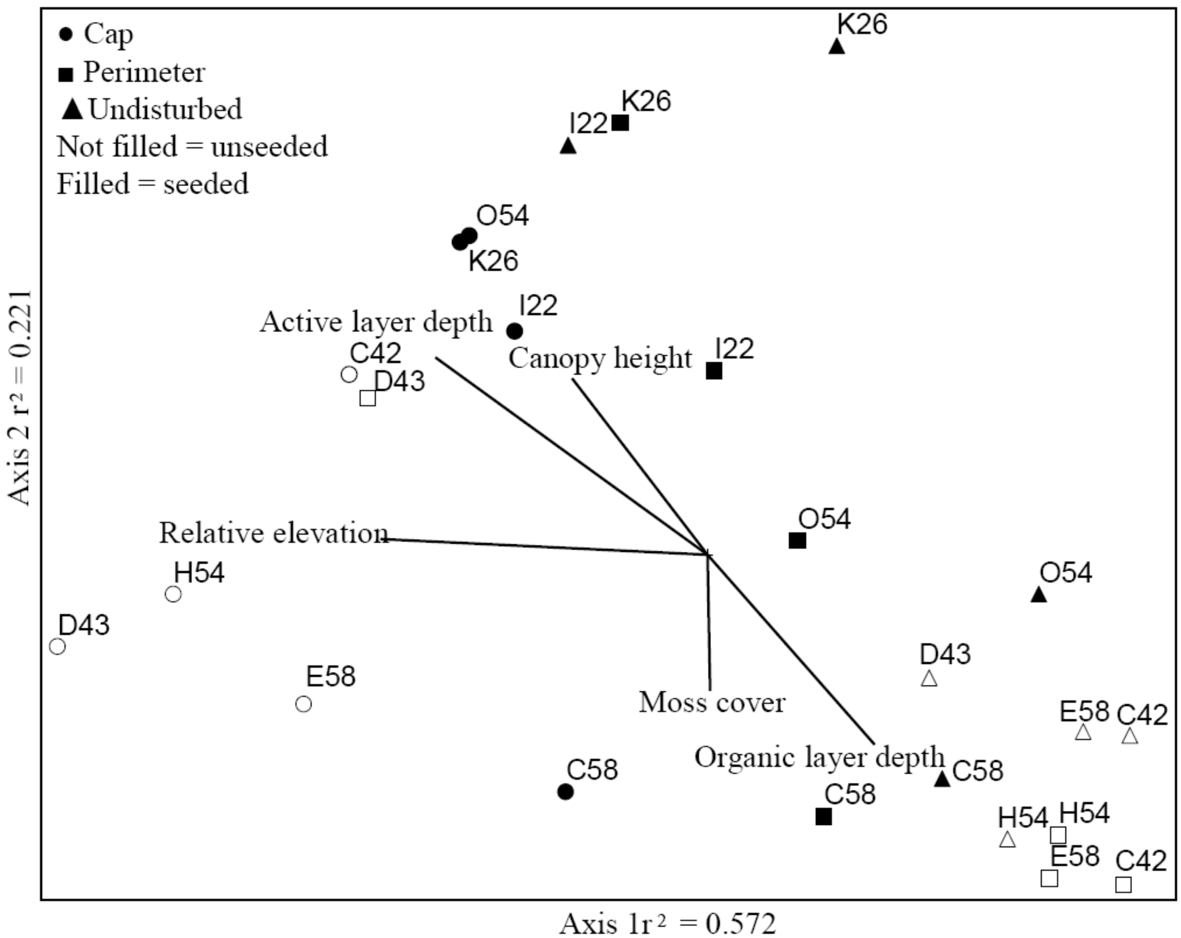
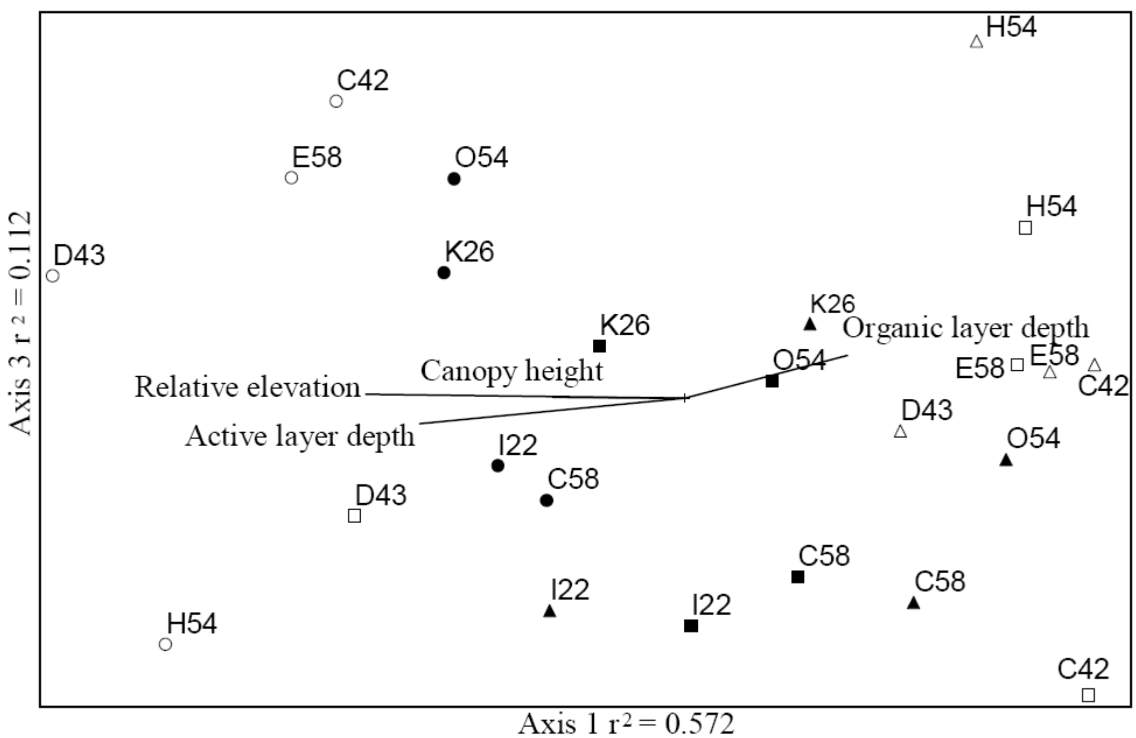

D43 cap
I22 und
I22 per
K26 cap
K26 per
I22 cap
O54 cap
K26 und
C42 cap
H54 cap
J06 cap
D43 per
C58 cap
O54 per
O54 und
D43 und
J06 per
E58 cap
K16 cap
C58 und
F09 per
C58 per
C42 und
F09 cap
D58 cap
E58 und
E58 per
C42 per
K16 per
H54 und
J06 und
F09 und
H54 per
D58 per
K16 und
D58 und
Axis 2 r 2 = 0.199
Axis 1 r 2 = 0.629
D43 cap
I22 per
I22 und
I22 cap
K26 cap
K26 per
K26 und
O54 cap
H54 cap
C42 cap
J06 cap
D43 per
J06 per
C58 cap
O54 per
D43 und
O54 und
E58 cap
K16 cap
C58 und
F09 per
C58 per
C42 und
E58 und
F09 cap
C42 per
E58 per
D58 cap
K16 per
H54 und
J06 und
H54 per
F09 und
D58 per
K16 und
D58 und
Axis 2 r 2 = 0.199
Axis 3 r 2 = 0.033
■ Lowland
▲ Upland
No fill = Unseeded
Solid fill = Seeded

Figure 3.2: (following page): Distribution of lowland sites with species frequency data of a three-dimensional NMS ordination capturing a total of 90.5 % variation. The top ordination is plotted on axis 1 and 2 and the bottom is plotted on axis 1 and 3. The NMS ordination used 200 iterations to produce a solution with a final stress of 9.97 and instability of 0.00006. The symbols indicate whether sites were unseeded (not filled) or seeded (filled) and located on the cap (circle), perimeter (square), or undisturbed zone (triangle). Line vectors indicate the strength and direction of correlations ($\tau \geq 0.232$) of axis scores with the labeled environmental variables organic layer depth (cm), moss cover (%), canopy height (cm), active layer depth (cm), and relative elevation (m).



● Cap
■ Perimeter
▲Undisturbed
Not filled = unseeded
Filled = seeded
Active layer depth
Canopy height
Relative elevation
Moss cover
Organic layer depth
K26
I22
K26
O54
K26
I22
C42
D43
I22
O54
H54
O54
D43
D43
E58
E58
C42
C58
C58
C58
H54
H54
E58
C42
Axis 2 r² = 0.221
Axis 1r² = 0.572



H54
C42
E58
O54
H54
D43
K26
K26
K26
Organic layer depth
O54
E58
E58
C42
Relative elevation
Canopy height
Active layer depth
D43
O54
I22
C58
D43
C58
C58
I22
I22
H54
C42
Axis 3 r² = 0.112
Axis 1 r² = 0.572

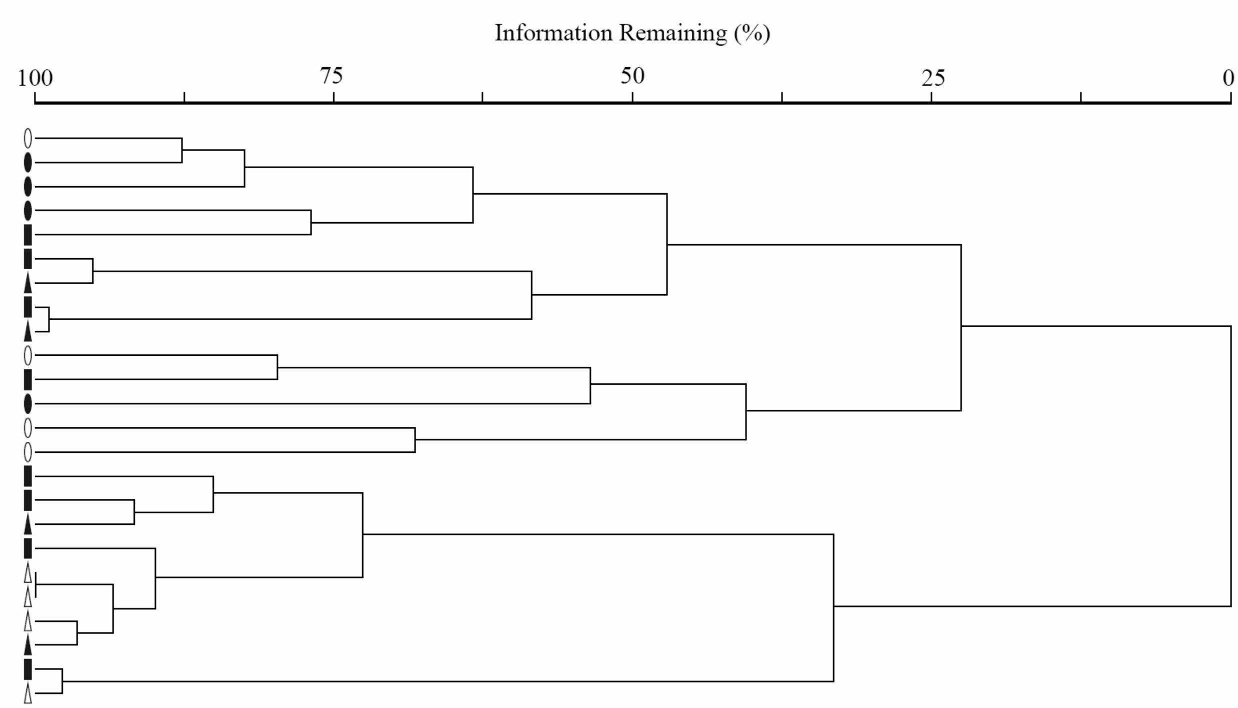


Figure 3.3: A dendrogram showing the hierarchical classification using a flexible beta algorithm of lowland sites divided into unseeded (not filled) and seeded (filled) sites located on the cap (circle), perimeter (square), or undisturbed zone (triangle).

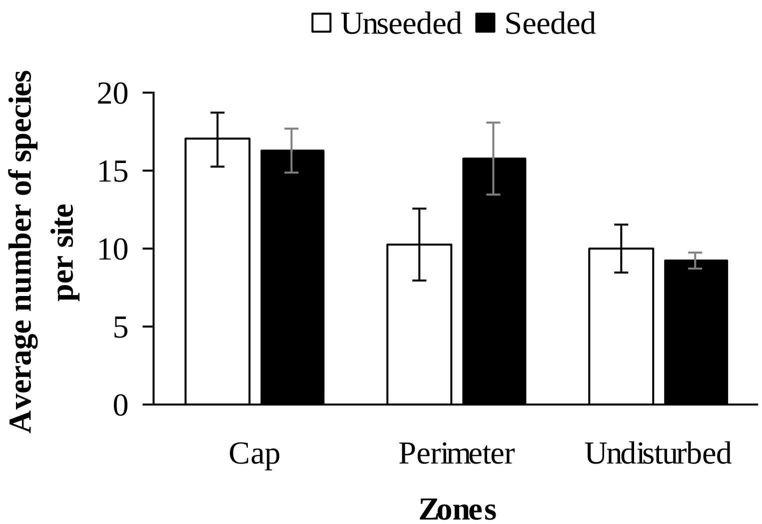


Figure 3.4: Mean species richness (number of species present per quadrat per site) for each zone of the sump (cap, perimeter, and undisturbed) divided into seeded and unseeded treatments in lowland terrain. The bars represent the standard error of the mean.

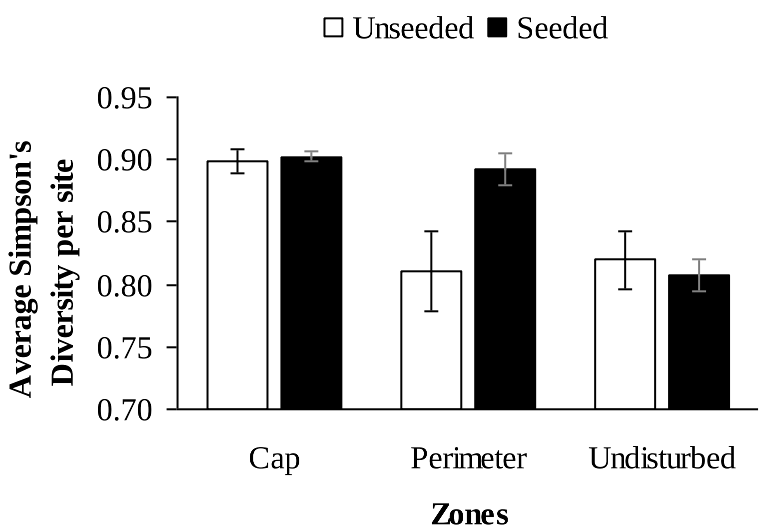


Figure 3.5: Mean Simpson's diversity for each zone of the sump (cap, perimeter, and undisturbed) divided into unseeded and seeded treatments in lowland terrain. The bars represent the standard error of the mean.

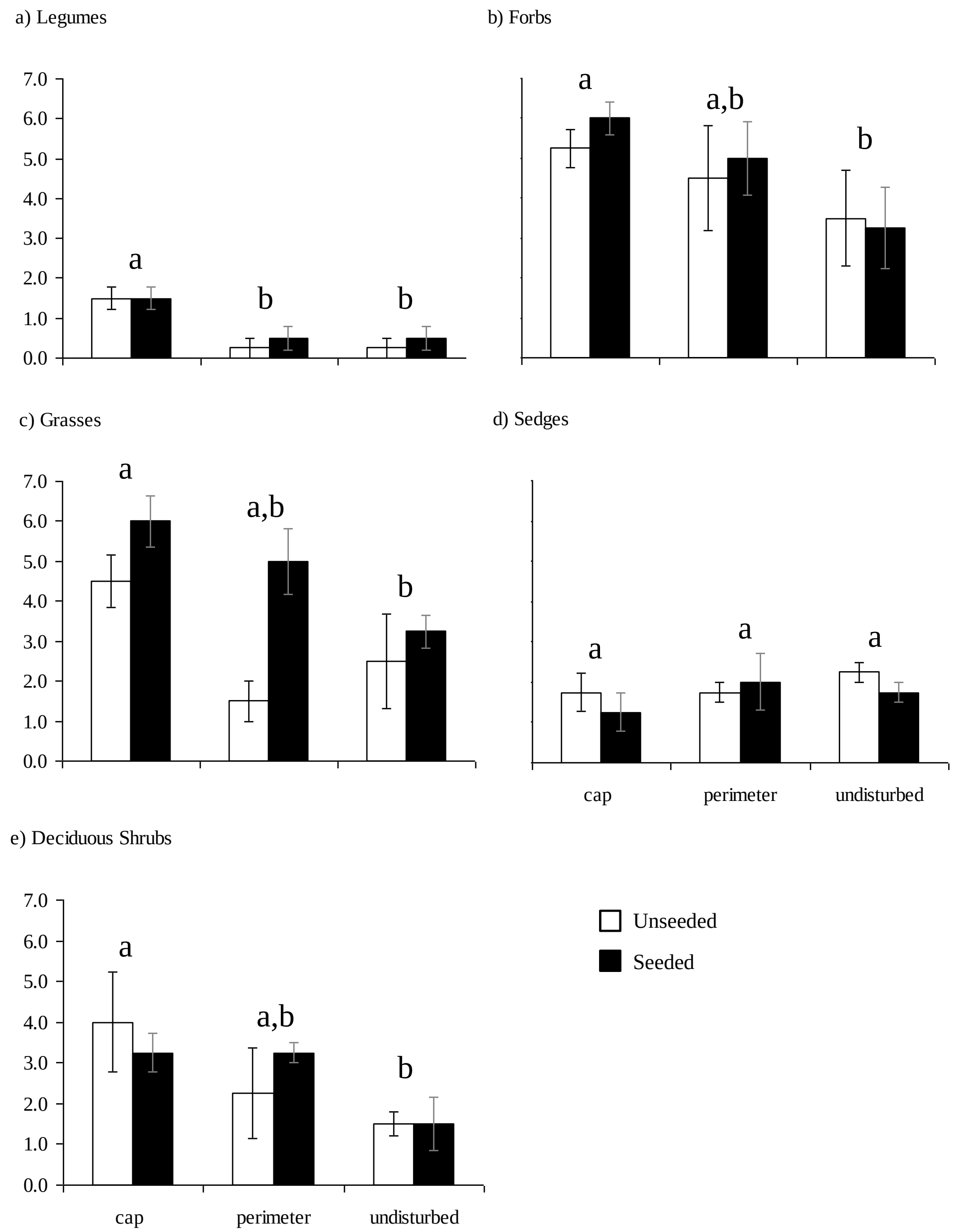


Figure 3.6: Average species richness of five plant functional types: a) legumes, b) non-legume forbs, c) grasses, d) sedges/rushes, and e) deciduous shrubs) found in the lowland terrain sites. Species richness is presented as the average (± 1 SE) across all of the sites (n = 8) for each of the three disturbance zones (cap, perimeter, and undisturbed). The sites are divided into unseeded (n = 4) and seeded treatments (n = 4), however statistical tests indicated that seeding treatment effects were non-significant. For a given variable, bars that share a letter indicate disturbance zones that are not significantly different from each other ($p < 0.05$) except deciduous shrubs ($p = 0.109$).

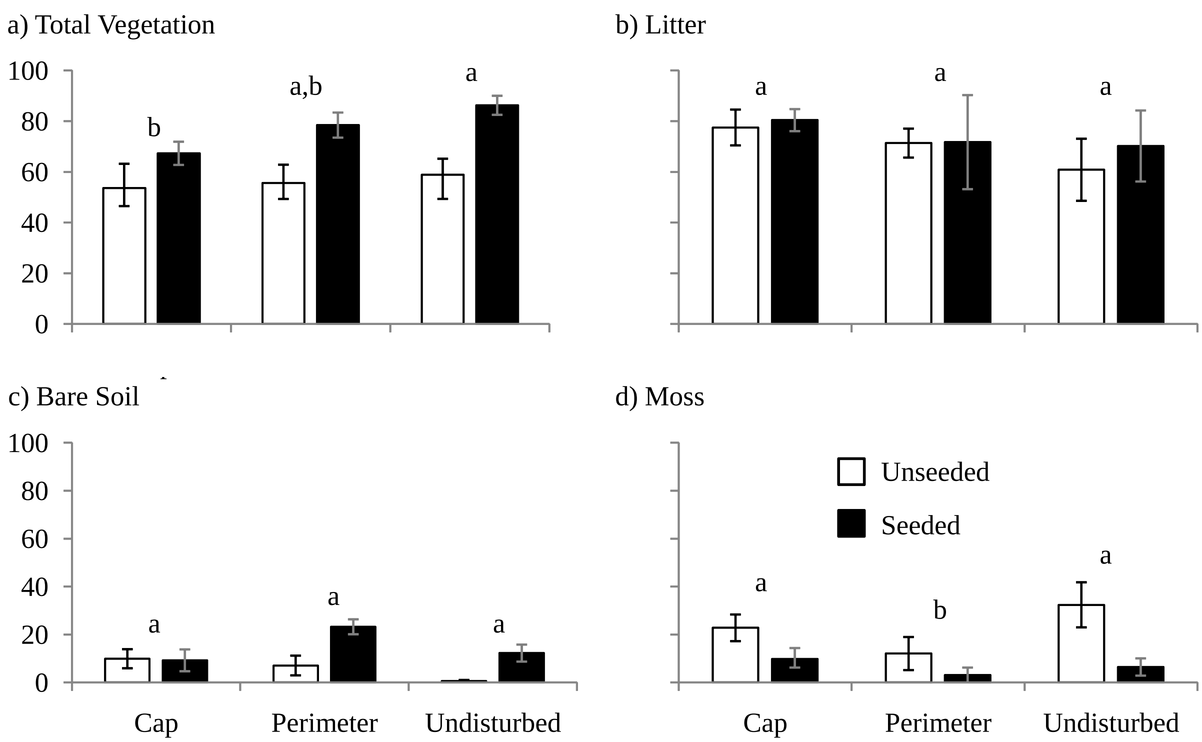


Figure 3.7: Average percent cover (± 1 SE) in different categories of surface cover (total vegetation, litter, bare soil, and moss) divided into the three disturbance zones (cap, perimeter, and undisturbed) and seeding treatments (unseeded n = 4; seeded n = 4). Seeding treatment was non-significant. For a given variable, bars that share a letter indicate disturbance zones that are not significantly different from each other (seeding treatments pooled = $p < 0.01$).The error bars represent standard error of the mean.

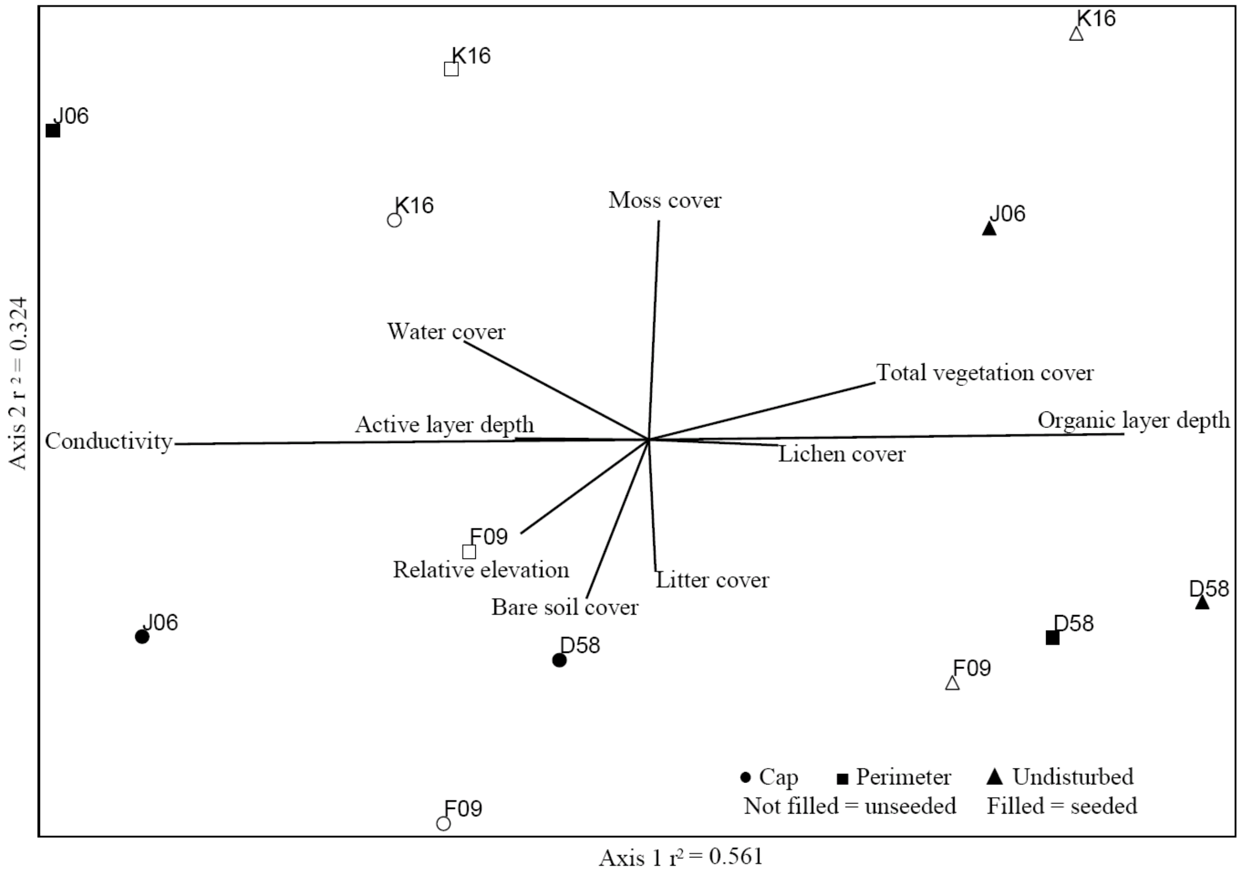


Figure 3.8: Distribution of upland sites with species proportion data plotted on the first and second ordination axes of a two-dimensional NMS ordination. The NMS ordination used 39 iterations to produce a solution with a final stress of 8.01 and instability of 0.0000001. The symbols indicate whether sites were unseeded (not filled) or seeded (filled) and located on the cap (circle), perimeter (square), or undisturbed zone (triangle). Line vectors indicate the strength and direction of correlations (τ>0.240) of axis scores with the labeled environmental variables total vegetation cover (%), water cover (%), lichen cover (%), bare soil cover (%), conductivity (dS/m), and organic layer depth (cm).

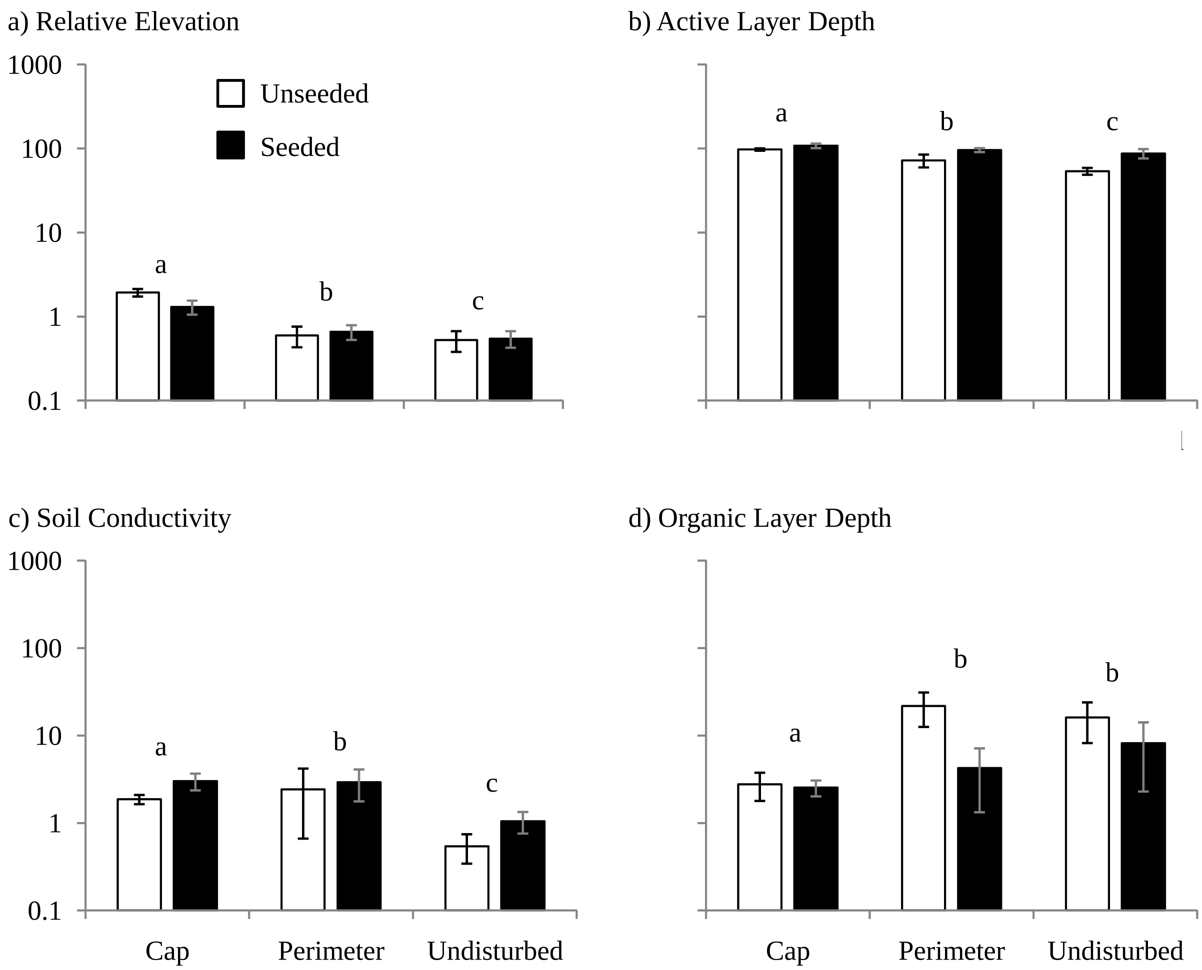


Figure 3.9: Average values (± 1 SE) of environmental variables (relative elevation, active layer depth, soil conductivity, and organic layer depth) plotted on a log axis across the three zones of the sump (cap, perimeter, and undisturbed) and seeding treatments (unseeded n = 4; seeded n = 4) for lowland sites. Seeding treatment was non-significant. For a given variable, bars that share a letter indicate disturbances zones that are not significantly different from each other (seeding treatments pooled = $p < 0.05$). Differences in organic layer depths between zones were marginally significant ($p = 0.078$)

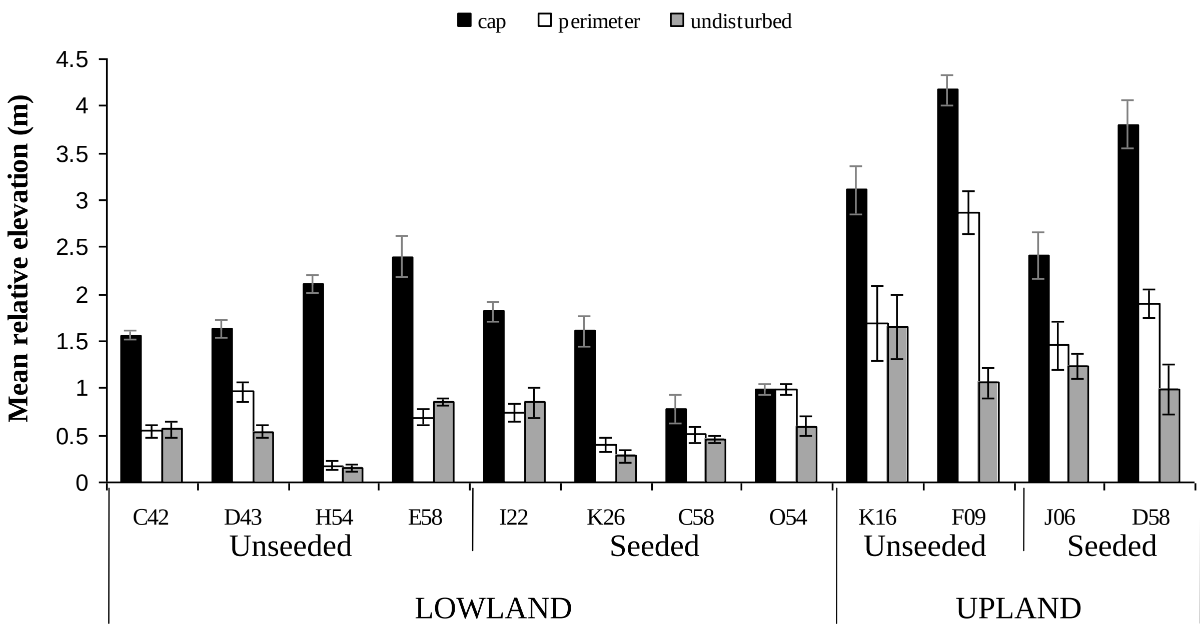


Figure 3.10: Relative elevation (site means ± 1 SE) for the sump sites in both lowland and upland terrain divided into the zones of the sump (cap, perimeter, and undisturbed) for unseeded and seeded treatments.

Figure 3.11: Photo of a sump cap eroding into a thermokarst pond in the perimeter of the sump at site D-43. Photo credit: Nicole Wunderlich, July 27, 2008.

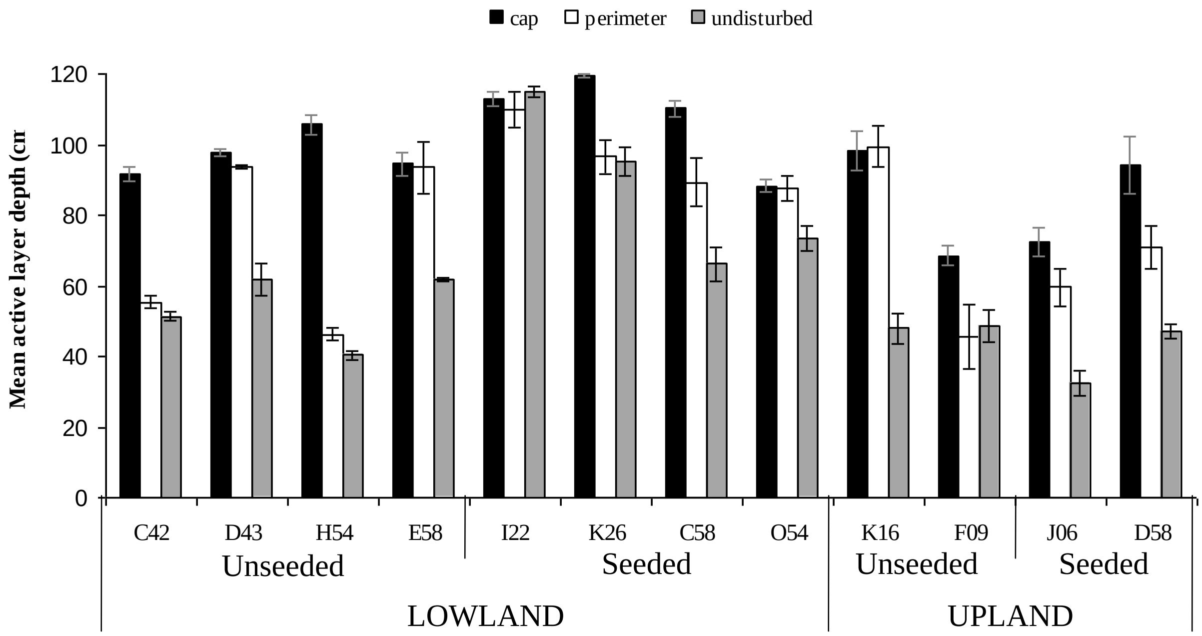


Figure 3.12: Active layer depths (cm; site means ± 1 SE) for both lowland and upland sites divided into the three zones of the sump (cap, perimeter, and undisturbed) for unseeded and seeded treatments.

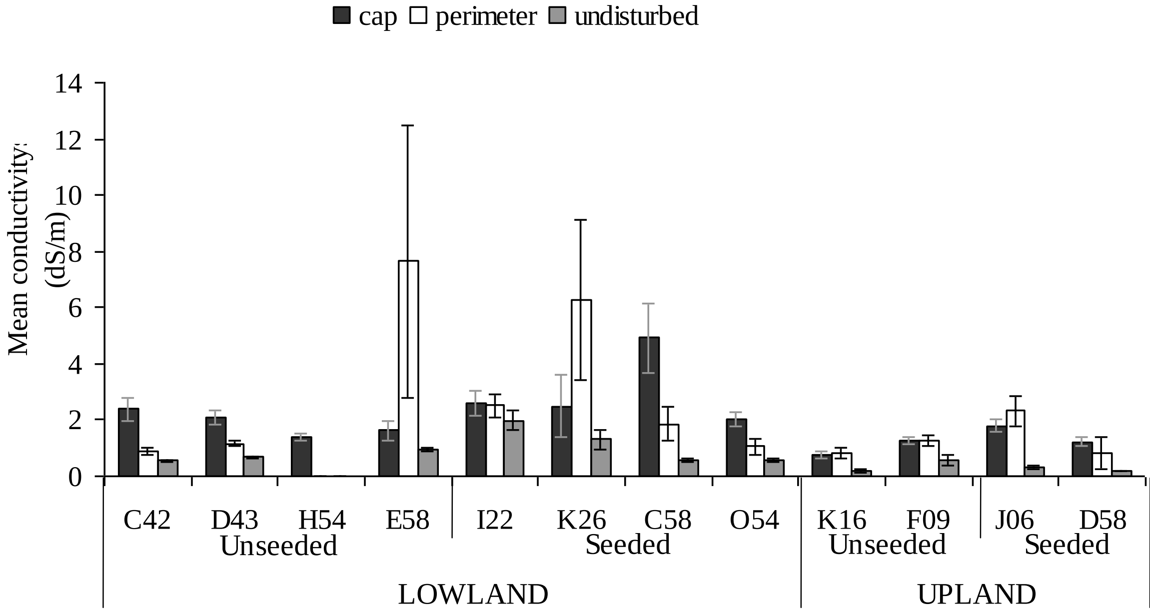


Figure 3.13: Soil conductivity (dS/m; site mean ± SE) averaged for both lowland and upland divided terrain and into the zones of the sump (cap, perimeter, and undisturbed) for both unseeded and seeded treatments.

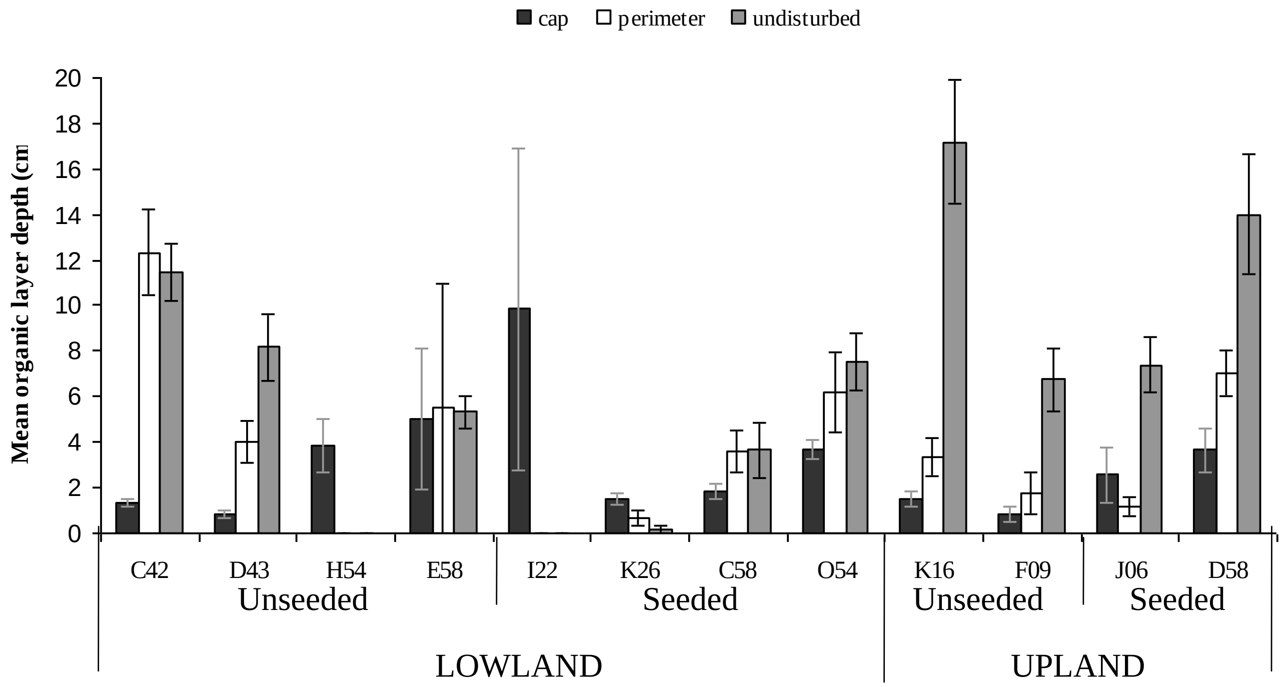


Figure 3.14: Organic layer depth (cm; site mean + SE) for both the lowland and upland terrain divided by the zones of the sump (cap, perimeter, and undisturbed) for unseeded and seeded treatments. Sites H-54 and E-58 had very wet marshy perimeter and undisturbed zones which meant the organic layer could not be properly measured. Sites I-22 and K-26 had evidence of frequent flooding (driftwood) and had layers of organic material within the mineral soil at times, but generally lacked an organic layer in the low-lying perimeter and undisturbed zones.

4. Discussion

4.1 Impacts of Seeding Treatments on Vegetation

After over 30 years of recovery, I found no significant effect of early seeding treatments on the plant community composition of drilling mud sumps in low arctic tundra of the Mackenzie River Delta. Seeding and fertilizing did not seem to affect the ability of native species to become established on the sumps. There was a persistent population of non-native grass species encountered on the caps and perimeters of the sumps at both seeded and unseeded sites, which may lead to invasive species management problems in the future. Introduced grass species *Poa pratensis* (Kentucky Bluegrass) and *Festuca rubra* (Creeping Red Fescue) were included in the seeding treatment by Younkin and Martens (1976). *F. rubra* was encountered more frequently on seeded sumps. However, their presence in the plant communities of sump caps cannot solely be attributed to the seeding treatment, as they were found at both seeded and unseeded sites. The sump caps may be vulnerable to invasion by non-native species because of the change in environment. In a different study, seeded species *F. rubra* cultivar "Arctared" and *P. pratensis* cultivar "Nugget" both invaded surrounding unseeded, disturbed areas from adjacent seeded patches 12 years after seeding (Younkin and Martens 1987). Introduced species such as *F. rubra* and *P. pratensis* could also become invasive in natural habitats that are similar to the sumps i.e. thaw slumps and point bars. *P. pratensis* cultivar "Nugget" is no longer recommended for standard revegetation because it can become a weed due to its overly aggressive spread by rhizomes (Hunt and Wright 2007, Wright 2008). This weedy habit may prevent native species re-establishment and thus *P. pratensis* has been recommended for use only in urban and residential areas (Hunt and Wright 2007, Wright 2008).

Other introduced species found on the sumps in this study included non-seeded species *Alopecurus pratensis* (Meadow Foxtail) and *Festuca trachyphylla* (Hard Fescue). These species were introduced to North America from Europe as a crop forage species (Cody 2000). However, *A. pratensis* and *F. trachyphylla* were not reported in the NWT until recently when they were found on the Norman Wells pipeline (Cody et al. 2000). Disturbance can increase the likelihood of invasion of an ecosystem especially since populations of these introduced species persist in the MRD (Hobbs and Huenneke 1992). Environments in the MRD and KIBS may change in the coming decades due to the direct effects of industrial disturbance as well as increased disturbance events such as flooding caused by extreme weather, and increased thermokarst because of warming temperatures (Billings 1997, Burn and Kokelj 2009, Morse et al. 2009). When managing disturbances we must consider the potential for invasion of non-native species as well as changes in the local environmental conditions that may have both direct impacts and synergistic effects with climate change or other environmental changes.

Even though there was no significant effect of seeding in the multivariate tests, there were interesting patterns between the unseeded and seeded sites in the ordination and cluster analysis for the lowland terrain (Figures 3.2 and 3.3). The sump caps of the seeded and unseeded sites seemed to have different vegetation communities, but this may be attributed to the site conditions and the integrity of the sump. The seeded site C-58 had a cap that was more similar to the unseeded caps and undisturbed zones because the sump had very little relief (Table 3.1 and Figure 3.10). Sites I-22, K-26, and O-54 (seeded) seemed to have a different plant community on the caps, but this may be due to the severe ponding and the collapsed/collapsing sumps at these sites instead of the seeding treatment (Table 3.1). Because many sump sites showed different, site-specific patterns of disturbance responses, the overall effects of the seeding treatment may have been masked by the variation between the sites. Nevertheless, the clear evidence of collapsing sump caps and water ponding at seeded sites I-22, K-26, and O-54 (Table 3.1) suggests that post-disturbance seeding treatments are not effective in preventing long-term patterns of sump erosion or collapse.

Although seeding did not have a significant effect on vegetation recovery in the long-term, revegetation treatments may still be desirable to help prevent erosion that may occur within a few years of disturbance. Special consideration should be taken when choosing future revegetation treatments in the arctic, particularly seeding with non-native species. Rehabilitation efforts must consider the both the short and long-term consequences of seeding with non-native species. There is evidence that seeding provides litter cover that may help in erosion control in the short-term (Younkin and Martens 1987). However, the data presented here indicate that non-native species can establish persistent populations in disturbed areas and seed applications of non-native species are likely to increase opportunities for non-native species to become established. In order to prevent possible spread of non-native species, seeding can be done with native cultivars, or indigenous species that were harvested from the surrounding undisturbed tundra by transplants, cuttings, or from seed. In addition, Gartner et al (1983) recommend the stockpiling and re-utilizing the soil organic layer for restoration practices in the arctic, which provides a source of native species propagules in the form of buried seeds and rhizomes. Alternative management practices could also involve adapting management plans over time in order to achieve desired results.

When interpreting the seeding treatment effects observed in this study, it is important to note that limitations in the study design may have influenced the ability to detect seeding effects. In particular, spatial variations between site locations in the Mackenzie Delta and unknown initial starting conditions may have confounded the comparison of seeding treatments. The sample size of the study was also relatively small compared to the high amount of variability in plant communities. Nevertheless, given the lack of information on decadal-scale effects of rehabilitation treatments on disturbance recovery, the results presented here are an important contribution to our understanding of seeding treatment effects.

4.2 Impacts of Sump Disturbances on Vegetation

The data collected in this study are consistent with previous research by Johnstone and Kokelj (2008), which showed a significant, long-term effect of sump disturbance on plant communities in the MRD. The initial disturbance of sump creation completely disrupted the plant community by removing vegetation and the soil organic layer and depositing an elevated cap of mineral soil. After 31 to 36 years of recovery, plant community composition on the sump caps was significantly different compared to the surrounding undisturbed tundra. Instead of the plant community on the sump resembling the original floristic community (represented here by undisturbed surrounding tundra), it has been replaced by a distinct community dominated by pioneer species. Indigenous species *Arctagrostis latifolia* and *Eriophorum angustifolium* subsp. *angustifolium* (Fireweed) were still abundant on the cap. Plant community composition on the sumps suggest that they are in the initial stages of succession (Svoboda and Henry 1987).

The creation of pioneer plant communities on the sump caps likely explains the higher species richness observed on sump caps in lowland terrain. This increased species richness in disturbed areas was attributed to an increase in species of legumes, non-legume forbs, grasses, and deciduous shrubs. The creation of un-vegetated sump caps provided an opportunity for pioneer species to colonize and spread, a process facilitated by less competition for light, space, and nutrients (Hernandez 1973). McKendrick et al. (1997) argue that an increase in plant species richness in disturbed areas may be due to the increased presence of herbaceous species. Harper and Kershaw (1996) examined the natural recovery of vegetation on borrow pits and vehicle tracks 48 years after pipeline construction in the NWT. They found significant differences in plant species composition with greater species richness on disturbed sites, but lower abundance of woody species compared to the undisturbed tundra (Harper and Kershaw 1996). These results are contrary to previous research on these sumps which found no difference in plant species richness between the zones (Johnstone and Kokelj 2008). This difference between our two studies could be because this study used a more even distribution of sampling across the zones and sampled almost twice as many sump sites compared to Johnstone and Kokelj (2008).

In contrast to patterns of species richness, the total vegetation cover was greatest in the undisturbed surrounding tundra in the lowland. Lower cover on the sump cap may be attributed to the slow rate of colonization or spread (Forbes and Jefferies 1999) and high levels of soil salinity that may be toxic or inhibit plant growth (Bliss and Svoboda 1984, Kokelj et al. 2002). Lowland terrain has been shown to have higher background solute concentrations (mostly sodium and calcium ions) due to annual flooding in comparison to upland tundra, which may make the upland more sensitive to increases in salinity (Kokelj and Burn 2005). The sump cap and perimeter zones were moderately saline in both the lowland and upland with several sites that had severely saline zones and bare soil patches devoid of vegetation. This could be due to drilling-waste (potassium chloride) leaking from the sump into the surrounding area. Potassium chloride (KCl) has been found to move up to 50 m laterally from the sump edges in both the lowland and upland terrain of the MRD (Kokelj and GeoNorth 2002). However, at the base of the active layer, it has been shown that there is a natural increase of solutes due to their exclusion during permafrost freezing (Kokelj and Burn 2005). Permafrost has a high concentration of solutes and thus with degradation, these ions will be released into the active layer (Kokelj and Burn 2005). Thus permafrost degradation can cause an increase in soil salinity, which can persist for centuries (Kokelj et al. 2002). The size of the disturbance and thus distance from undisturbed vegetation may have contributed to the lower total vegetation cover on the sump caps. Many plant species of the low arctic tundra can colonize neighbouring disturbances by asexual reproduction, including *Carex* and *Eriophorum* species that produce clones from rhizomes. However, when the disturbance is greater than 1-2 m^2, aerial seed dispersal or a seed bank is required for colonization (Forbes and Jefferies 1999). This may explain why the perimeter zone had a more similar vegetation composition to the undisturbed zone.

In upland terrain, species richness of vascular plants was lower on the sump caps compared to undisturbed tundra, largely due to the lack of evergreen shrubs. Evergreen shrubs, including ericaceous species (Ericaceae and Empetraceae), are sensitive to disturbance (Bliss and Wein 1972, Hernandez 1973, Forbes et al. 2001). Sump caps in upland tundra were relatively depauperate of vegetation, but had a high abundance and frequency of grasses and *Epilobium angustifolium* subsp. *angustifolium* (Fireweed). Contrary to my results, Kemper and Macdonald (2009) found that species richness in upland tundra disturbed by seismic lines 20 to 30 years previously in KIBS was not significantly different from the undisturbed tundra. However, they also found that seismic lines had a higher cover of vascular plants compared to the undisturbed tundra because of an increased cover of grasses and deciduous shrubs on the seismic lines (Kemper and Macdonald 2009). In my study the undisturbed upland plant community also had high lichen cover representative of native shrub-heath tundra (Hernandez 1973, Forbes et al. 2001). Lichens were not found on the sump caps and perimeters because lichens are sensitive to disturbance (Hernandez 1973, Kemper 2005).

The plant community on the sump cap resembles that of natural disturbances in the MRD region, such as thaw slumps and point bars. Seed sources of many sump colonizers are likely from natural disturbances. These natural disturbance communities are not stable, but are dynamic because they are in different stages of succession (Raynolds and Walker 2009). Sump caps in the lowland terrain may be similar in composition to river alluvium communities such as point bars because they are largely dry, but experience occasional flooding (Gill 1972, Johnstone and Kokelj 2008). River bars follow a successional pattern from tall shrubs toward dwarf shrub or herbaceous tundra (Bliss and Cantlon 1957). River bar communities are initially dominated by tall shrubs including *Salix alaxensis* (Gill 1972). The presence of these tall shrubs on the point bars warms the soil and can lead to permafrost degradation (Smith 1975). The sump may be similar in this respect, as *S. alaxensis* occurred at high frequency and abundance on several sump caps. The sump disturbances also have some resemblance to thaw slumps because of the degradation of permafrost. Thaw slumps are natural disturbances in the tundra where permafrost degrades causing a wall of soil to slide down the side of a hill leaving a large area of disturbed soil and exposed permafrost (Lantz et al. 2009). The presence of tall shrubs, absence of an organic layer, low albedo, and a thick active layer are common characteristics of both thaw slumps and sump caps (Johnstone and Kokelj 2008, Lantz et al. 2009).

Sumps create a unique local habitat for colonizing species. This contributes to forming a distinct vegetation community on the sump cap. This habitat was characterized by changes in the environment that include an increase in elevation, active layer depth, and decrease in organic layer depth. The sumps are elevated above the undisturbed tundra, and distinct vegetation communities on the sump caps will likely persist as long as the elevation difference persists (Johnstone and Kokelj 2008). Increased elevation has been shown to cause a decrease in moisture (increased drainage) that is thought to drive the differences in plant community composition (Lawson et al. 1978, Forbes et al. 2001, Johnstone and Kokelj 2008). Increases in the active layer may lead to subsidence, thermokarst ponding, and the leakage of drilling-waste (Kokelj and GeoNorth 2002).

4.3 Ecosystem Implications of Altered Vegetation

Arctic ecosystems are sensitive to disturbance because of the presence of permafrost (Bliss et al. 1973), which can degrade if the vegetation is removed or even compacted (Bliss et al. 1973, Burn and Kokelj 2009). The ground ice content in permafrost dictates the ecological sensitivity of the terrain (Burn and Kokelj 2009). In the MRD, ice wedges indicate that there is a high ground ice content in the top five meters of permafrost in both lowlands and uplands (Burn and Kokelj 2009). Creation of a sump and subsequent capping can cause changes in the underlying ice-rich permafrost. When the sump was created, the initial surface disturbance (i.e. removal of plant material, organic layer, and exposure of mineral soil and permafrost) may have led to degradation of permafrost, especially if the sump was open during the summer (Kokelj and GeoNorth 2002). Once the sump was capped (after the well was decommissioned) the permafrost may not recover fully, leading to an increase in active layer depth (Kokelj and GeoNorth 2002, Lantz et al. 2009). This can lead to the leakage of drilling-wastes through the active layer (Kokelj and GeoNorth 2002).

The presence of tall shrubs on sump caps has been shown to affect the snow accumulation in the winter and may contribute to sump degradation (Sturm et al. 2005, Johnstone and Kokelj 2008). Tall shrubs, such as *Salix* and *Alnus* species, trap the snow and are associated with increased snow depth on the sump cap (Johnstone and Kokelj 2008). This increase in snow depth can cause an increase in the temperature of the soil as well as the near surface permafrost (Sturm et al. 2001, Sturm et al. 2005, Burn and Kokelj 2009). With increasing snow depth, the soil temperature increase can cause thermokarst erosion (Hinkel and Hurd 2006). This may lead to increased ponding around the sump and eventual sump subsidence.

4.4 Management Implications

The long-term effect of introducing non-native species during revegetation has implications for management of disturbances in the Arctic. Revegetation is one mitigation strategy that has been used inconsistently in the past, and little is known about its long-term effects. The initial, short-term goals of Younkin and Martens' (1976) revegetation treatment were to prevent erosion, minimize the effects of thermokarst, and where possible restore the tundra ecosystem to its natural state. Younkin and Martens (1979) argued that the litter produced by seeded species aided in erosion prevention. In the long-term, seeding with non-native species may increase the risk of species invasion on the sumps and in similar natural disturbances. It is important to recognize this risk and work towards preventing species invasion in the Arctic. Therefore revegetation strategies must consider both the short and long-term effects of revegetation treatments (Forbes et al. 2001). If revegetation is necessary in order to prevent erosion in the short-term, alternative strategies should be considered for the Arctic. These strategies may include site preparation (e.g. removing gravel), seeding with indigenous species, seeding with a variety of plant functional types including forbs (e.g. nitrogen fixing legumes), and monitoring recovery in order to reduce the possibility of introducing non-native species.

Sump integrity may have been driving the differences observed here between seeded and unseeded sites in the ordinations and cluster analysis. Several seeded sumps were collapsing and had ponding in the perimeter that may have contributed to differences in the vegetation communities between these treatments. Sump creation and abandonment practices affect the integrity of the sumps and may be more important to the recovery of the sump sites than seeding treatment in erosion control. Single season, winter exploration followed by decommissioning sumps using ample overburden has been associated with improved sump functioning (Kokelj and GeoNorth 2002).

The creation of elevated structures, such as sumps or buried pipelines with berms is not recommended for future projects in the Arctic. The change in elevation (~ 1 m) on the tundra has a long-term effect on both the plant community and the environment as seen in this study and in Johnstone and Kokelj (2008). Sumps in the MRD also have many problems including drilling-waste leakage and thermokarst subsidence causing sump collapse and ponding (Kokelj and GeoNorth 2002). Half of the sump sites I visited had caps that were actively slumping. Most sites had ponding in the perimeter (11 out of 12), which was evidence of thermokarst erosion. Problems with sumps varied and the differences from site to site were probably due to the drilling practices, including sump location and rehabilitation (Kokelj and GeoNorth Ltd 2002, AMEC Earth and Environment 2004). For example, drilling in the summer leads to increased subsidence, larger sumps, and ponding (Kokelj and GeoNorth 2002). Ponds are a heat sink and if they are adjacent to sumps they usually result in degradation of the sump (Kokelj and GeoNorth 2002). Single season, winter operations were less destructive because the drilling-waste remains frozen and ice roads reduce the chances of vegetation being severely trampled and disturbed (French 1980). In the context of climate change, sumps may not be practical for areas with near-surface permafrost temperatures close to thawing thresholds that can eliminate permafrost in a warming climate (Burn and Kokelj 2009, Morse et al. 2009). These results suggest that perhaps sumps, as a way to dispose of drilling-wastes, should not be used in the MRD. Other options for drilling-waste disposal include a closed loop drilling-fluid system involving reuse of drilling fluids, zero waste discharge, or deep disposal wells (Serverson-Baker 2004).

Understanding of long-term effects of these sump disturbances is important for assessing cumulative impacts in KIBS both before and after new industrial development. Taking the average disturbance footprint of a sump (cap and perimeter) from this study and multiplying it by the 19 wells in KIBS, the total area disturbed by sumps is an estimated 0.34 km^2 of the land area in KIBS. This is a conservative estimate because some well sites have more than one sump. Currently, there is a greater than 1 % footprint from anthropogenic disturbance in KIBS, which may have a negative impact on the migratory birds because the undisturbed tundra plant community is important for wildlife habitat, stabilizing the soil, maintaining permafrost, and it functions as a carbon sink (Schindler and Smol 2006). Increasing industrial development in KIBS can lead to habitat loss, reduced habitat quality, and the alteration of nesting and foraging habitats for migratory birds (Ashenhurst 2004).Assessment of cumulative impacts of industrial development should explicitly consider that many disturbances leave behind a long-term legacy of habitat alteration.

There are many areas for further research on disturbances in the Arctic. These revegetation studies should continue to be monitored in the future in order to gain longer-term effects on vegetation recovery. Information is lacking or not readily available on current anthropogenic disturbances in the Canadian Arctic and the recovery of past disturbances. Moreover, there is a need to investigate the spread of introduced and possibly invasive species into the native tundra and on natural disturbances such as thaw slumps and river bars. It is particularly important to investigate the responses of seeded species in the context of climate warming.

5. Conclusions

This study found no significant effect of seeding treatments applied in 1975-76 on either the plant community composition, diversity, or environmental variables measured in 2008. This suggests that the low arctic tundra of the MRD may currently be resistant to the maintenance or spread of introduced non-native species. However, the potential for invasion and spread of non-natives may increase with climate warming and increases in land disturbance due to natural causes or human activities. Sump disturbances could act as centers for the spread of non-native species. Introduced grass species were encountered on the caps and perimeters of the sumps at both seeded and unseeded sites including *Poa pratensis* and *Festuca rubra*, which were part of the seeding treatment by Younkin and Martens (1976).

The plant communities on sumps have not recovered after over 30 years, indicating that these disturbances represent a long-term impact on plant communities. These results should be considered for future management in KIBS, including assessments of the cumulative impacts of industrial disturbance. The creation of elevated structures such as sumps has a long-term effect on both the plant community and the environment. Sumps may not be suitable for areas with ice-rich near surface permafrost, especially in the context of climate warming in the North because of the possibility for permafrost melting that can cause sump subsidence and leakage of drilling-waste.

6. References

Alaska Biological Research Inc. 2006. Rehabilitation report for the MS3 trenching site rehabilitation project, North Slope, Alaska. Report, Environmental Research and Services, Fairbanks, Alaksa and BP Environmental Studies Group, Anchorage, Alaska.

AMEC Earth and Environment. 2004. Inuvialuit settlement region drilling waste disposal sumps study. Government of Canada Environmental Studies Research Funds, Calgary, AB.

Anisimov, O. A., D. G. Vaughan, T. V. Callaghan, C. Furgal, H. Marchant, T. D. Prowse, H. Vilhjálmsson, and J. E. Walsh. 2007. Polar regions (Arctic and Antarctic). Pages 653-685 *in* M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden, and C. E. Hanson, editors. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge.

Ashenhurst, A. R. 2004. The effects of seismic lines and drill pads on breeding migratory birds in the Kendall Island Migratory Bird Sanctuary, NWT. University of Alberta, Edmonton, AB.

Berger, T. R. 1977. Northern frontier, northern homeland: The report of the Mackenzie Valley Pipeline Inquiry. Supply and Services Canada, Ottawa, ON.

Billings, W. D. 1997. Challenges for the future: Arctic and alpine ecosystems in a changing world. Pages 1-18 Ecological Studies; Global change and Arctic terrestrial ecosystems. Springer-Verlag, New York, NY.

Bliss, L. C. 1983. Modern human impacts in the arctic.*in* M. J. A. Holzner and I. Ikusimal, editors. Man's Impact on Vegetation. Dr. W. Junk Publishers, The Hague.

Bliss, L. C. 1988. Arctic tundra and polar desert biome.*in* M. G. Barbour and W. D. Billings, editors. North American Terrestrial Vegetation. Cambridge University Press, New York, NY.

Bliss, L. C. and J. E. Cantlon. 1957. Succession on river alluvium in northern Alaska. American Midland Naturalist Journal **58**:452-469.

Bliss, L. C., G. M. Courtin, D. L. Pattie, R. R. Riewe, D. W. A. Whitfield, and P. Widden. 1973. Arctic tundra ecosystems. Annual Review of Ecology, Evolution, and Systematics **4**:359-399.

Bliss, L. C. and N. V. Matveyeva. 1992. Circumpolar arctic vegetation. Pages 59-89 *in* F. S. Chapin III, R. L. Jefferies, J. F. Reynolds, G. S. Shaver, and J. Svoboda, editors. Arctic Ecosystems in a Changing Climate: An Ecophysiological Perspective. Academic Press, San Diego, CA.

Bliss, L. C. and K. M. Peterson. 1992. Plant succession, competition, and physiological constraints of species in the Arctic. Pages 111-133 *in* F. S. Chapin III, R. L. Jefferies, J. F. Reynolds, G. R. Shaver, and J. Svoboda, editors. Arctic Ecosystems in a Changing Climate: an Ecophysiological Perspective. Academic Press, San Diego, CA.

Bliss, L. C. and J. Svoboda. 1984. Plant communities and plant production in the Western Queen Elizabeth Islands. Holarctic Ecology **7**:325-344.

Bliss, L. C. and R. W. Wein. 1972. Plant community responses to disturbances in the western Canadian Arctic. Arctic Canadian Journal Botany **50**:1097-1109.

Braun-Blanquet, J. 1932. Plant Sociology: The Study of Plant Communities. 1 edition. McGraw-Hill Book Company, Inc., New York, NY.

Bret-Harte, M. S., M. C. Mack, G. R. Goldsmith, D. B. Sloan, J. DeMarco, G. R. Shaver, P. M. Ray, Z. Biesinger, and F. S. Chapin, III. 2008. Plant functional types do not predict biomass responses to removal and fertilization in Alaskan tussock tundra. Journal of Ecology **96**:713-726.

Burn, C. R. and S. V. Kokelj. 2009. The environment and permafrost of the Mackenzie Delta area. Permafrost and Periglacial Processes **20**:83-105.

Canadian Food Inspection Agency. 2008. Invasive Alien Plants in Canada - Summary Report. Public Affairs, Nepean, ON.

Canadian Wildlife Service. 2004. A management agreement for the Kendall Island Migratory Bird Sanctuary. Environment Canada, Yellowknife, NWT.

Cargill, S. M. and F. S. Chapin III. 1987. Application of successional theory to tundra restoration: A review. Arctic and alpine research **19**:366-372.

Chapin, F. S., III. 1993. Functional role of growth forms in ecosystem and global processes. Pages 287-312 *in* J. R. Ehleringer, Field, C. B., editor. Scaling Physiological Processes: Leaf to Globe. Academic Press Inc, San Diego, CA.

Chapin, F. S., III, M. S. Bret-Harte, S. E. Hobbie, and H. L. Zhong. 1996. Plant functional types as predictors of transient responses of arctic vegetation to global change. Journal of Vegetation Science **7**:347-358.

Chapin, F. S., III and M. C. Chapin. 1980. Revegetation of an arctic disturbed site by native tundra species. The Journal of Applied Ecology **17**:449-456.

Clarke, K. R., P. J. Somerfield, and M. G. Chapman. 2006. On resemblance measures for ecological studies, including taxonomic dissimilarities and a zero-adjusted Bray–Curtis coefficient for denuded assemblages. Journal of experimental marine biology and ecology **330**:55.

Cody, W. J. 2000. Flora of the Yukon Territory. 2nd edition. NRC Monograph Publishing Program, Ottawa, ON.

Cody, W. J., K. L. MacInnes, J. Cayouette, and S. Darbyshire. 2000. Alien and invasive native vascular plants along the Norman Wells pipeline, District of Mackenzie, Northwest Territories. Canadian Field Naturalist **114**:126-137.

Corwin, D. L. and S. M. Lesch. 2003. Application of soil electrical conductivity to precision agriculture theory, principles, and guidelines. Agronomy Journal **95**:455-471.

Deshaies, A., S. Boudreau, and K. Harper. 2009. Assisted revegetation in a subarctic environment: Effects of fertilization on the performance of three indigenous plant species. Arctic, Antarctic, and Alpine Research **41**:434-441.

Dufrene, M. and P. Legendre. 1997. Species assemblages and indicator species: The need for a flexible asymmetrical approach. Ecological Monographs **67**:345-366.

Dunn, R. 2001. Salt tolerance of plants. AGRI FACTS, Calgary, AB.

Environment Canada. 2009. Canadian Climate Normals or Averages 1971-2000: Inuvik, Northwest Territories. National Climate Data and Information Archive.

Evans, K. E. and G. Kershaw. 1989. Productivity of agronomic and native plants under various fetilizer and seed application rates on a simulated transport corridor, Fort Norman, Northwest Territories. Pages 279-287 *in* Reclamation, A Global Perspecitve. Alberta Land Conservation and Reclamation Council, Calgary, AB.

Everett, K. R., B. M. Murray, D. F. J. A. W. Murray, A. E. Lintins, and P. J. Webber. 1985. Reconnaissance observations of long-term natural revegetation recovery in the Cape

Thompson region, Alaska and additions to the checklist of flora. U.S. Army Cold Region Research and Engineering Laboratory, Hanover, NH.

Faith, D. P., P. R. Minchin, and L. Belbin. 1987. Compositional dissimilarity as a robust measure of ecological distance. Vegetatio **69**:57-68.

Forbes, B. C. 1993. Small scale wetland restoration in the High Arctic: A long-term perspective. Restoration Ecology **1**:59-68.

Forbes, B. C. 1997. Tundra disturbance studies. IV. Species establishment in rubble and sand quarries, Yamal Region, Northwest Siberia, Russia. Polar Geography **21**:79-100.

Forbes, B. C., J. J. Ebersole, and B. Strandberg. 2001. Anthropogenic disturbance and patch dynamics in circumpolar arctic ecosystems. Conservation Biology **15**:954-969.

Forbes, B. C. and R. L. Jefferies. 1999. Revegetation of disturbed arctic sites: Constraints and applications. Biological Conservation **88**:15-24.

French, H. M. 1980. Terrain, land-use and waste drilling-fluid disposal problems, Arctic Canada. Arctic **33**:794-806.

Gill, D. 1972. Point bar environment in the Mackenzie River Delta. Canadian Journal of Earth Sciences **9**:1382-1393.

GNWT. 2009. Government of the Northwest Territories: NWT Species Infobase. http://www.enr.gov.nt.ca. Environment and Natural Resources Government of the Northwest Territories, Yellowknife, NWT.

Goetz, S. J., A. G. Bunn, G. J. Fiske, and R. A. Houghton. 2005. Satellite-observed photosynthetic trends across boreal North America associated with climate and fire disturbance. Proceedings of the National Academy of Sciences of the United States of America **102**:13521-13525.

Government of Canada. 2009. Committee on the status of endangered wildlife in Canada (COSEWIC). http://www.cosewic.gc.ca/eng/sct1/searchform_e.cfm.

Green, R. H. 1979. Sampling design and statistical methods for environmental biologists. John Wiley and Sons Inc., New York, NY.

Haag, R. W. and L. C. Bliss. 1974. Energy budget changes following surface disturbance to upland tundra. Journal of Applied Ecology **11**:355-374.

Harper, K. A. and G. P. Kershaw. 1996. Natural revegetation on borrow pits and vehicle tracks in shrub tundra, 48 years following CANOL No. 1 Pipleline, N.W.T., Canada. Arctic and alpine research **28**:163-171.

Hassol, S. J. 2004. Impacts of a Warming Arctic: Arctic Climate Impact Assessment. Cambridge University Press.

Hernandez, H. 1973. Natural plant recolonization of surficial disturbances, Tuktoyaktuk Peninsula Region, NWT. The Canadian Journal of Botany **51**:2177-2195.

Hinkel, K. M. and J. K. J. Hurd. 2006. Permafrost destabilization and thermokarst following snow fence installation, Barrow, Alaska, USA. Arctic, Antarctic, and Alpine Research **38**:530-539.

Hirst, C. N. and D. A. Jackson. 2007. Reconstructing community relationships: The impact of sampling error, ordination approach, and gradient length. Diversity and Distributions **13**:361-371.

Hobbs, R. J. and L. F. Huenneke. 1992. Disturbance, diversity, and invasion: Implications for conservation. Conservation Biology **6**:324-337.

Hunt, P. and S. Wright. 2007. 'Nugget' Kentucky Bluegrass.*in* Alaska Plant Materials Center, editor. Department of Natural Resources, Palmer, AK.

Huntington, H. P., M. Boyle, G. E. Flowers, J. W. Weatherly, L. C. Hamilton, L. Hinzman, C. Gerlach, R. Zulueta, C. Nicolson, and J. Overpeck. 2007. The influence of human activity in the Arctic on climate and climate impacts. Climatic Change **82**:77-92.

ITIS. 2009. Integrated Taxonomical Information System (ITIS) online database. http://www.itis.gov/index.html.

Johnson, L. A. 1981. Revegetation and selected terrain disturbance along the Trans-Alaskan pipeline. Cold Regions Research and Engineering Lab.

Johnstone, J. and S. Kokelj. 2008. Environmental conditions and vegetation recovery at abandoned drilling mud sumps in the Mackenzie Delta region, Northwest Territories, Canada. Arctic **61**:199-211.

Jorgenson, M. T. 1997. Patterns and rates of factors affecting natural recover on land disturbed by oil development in Arctic Alaska. Pages 421-422 *in* R. M. M. Crawford, editor. Disturbance and Recovery in Arctic Lands. Kluwer Academic Publishers, Netherlands.

Jorgenson, M. T. and M. R. Joyce. 1994. Six strategies for rehabilitation land disturbed by oil development in Arctic Alaska. Arctic **47**:374-390.

Jorgenson, M. T., J. G. Kidd, T. C. Carter, S. Bishop, C. H. Racine, R. O. Rasmussen, and N. E. Koroleva. 2003. Long-term evaluation of methods for rehabilitation of lands disturbed by industrial development in the Arctic. Pages 173-190 *in* R. O. Rasmussen and N. E. Koroleva, editors. Social and Environmental Impacts in the North: Methods in evaluation of socio-economic and environmental consequences of mining and energy production in the Arctic and Subarctic Kluwer Academic Publishers, Netherlands.

Kemper, J. 2005. Short and long-term effects of winter seismic exploration on low arctic plant communities of the Kendall Island Migratory Bird Sanctuary, NWT. Master's Thesis. University of Alberta, Edmonton, Canada.

Kemper, J. T. and S. E. Macdonald. 2009. Directional change in upland tundra plant communities 20-30 years after seismic exploration in the Canadian low-arctic. Journal of Vegetation Science **20**:557-567.

Kershaw, G. P. and L. J. Kershaw. 1987. Successful plant colonizers on disturbances in tundra areas of northwestern Canada. Arctic and Alpine Research **19**:451-460.

Kidd, J. G., B. Streever, and M. T. Jorgenson. 2006. Site characteristics and plant community development following partial gravel removal in an Arctic oilfield. Arctic, Antarctic and Alpine Research **38**:384-393.

Kokelj, S. and GeoNorth Ltd. 2002. Drilling mud sumps in the Mackenzie Delta region: Construction, abandonment and past performance. Department of Indian Affairs and Northern Development, NWT Region.

Kokelj, S., C. A. S. Smith, and C. R. Burn. 2002. Physical and chemical characteristics of the active layer and permafrost, Herschel Island, Western Arctic Coast, Canada. Permafrost and Periglacial Processes **13**:171-185.

Kokelj, S. V. and C. R. Burn. 2005. Near-surface ground ice in sediments of the Mackenzie Delta, Northwest Territories, Canada. Permafrost and Periglacial Processes **16**:291-303.

Kruskal, J. B. 1964. Nonmetric multidimensional scaling: A numerical method. Psychometrika **29**:115-129.

Lantz, T., J. Kanigan, S. Smith, R. Coutts, and S. Kokelj. 2009. Origin and polycyclic behaviour of tundra thaw slumps, Mackenzie Delta region, Northwest Territories, Canada. Permafrost and Periglacial Processes **20**:173-184.

Lawson, D. E., J. Brown, K. R. Everett, A. W. Johnson, and V. Komarkova. 1978. Tundra disturbances and recovery following the 1949 exploratory drilling, Fish Creek, northern Alaska.

Looman, K. and J. F. Best. 1979. Budd's Flora of the Canadian Prairie Provinces. Canadian Government Publishing Center, Hull, QC.

Mackay, J. R. 1963. The Mackenzie Delta Area, NWT. Geographical Branch, Mines and Technical Surveys, Ottawa, ON.

Mackay, J. R. 1970. Disturbances to the tundra and forest tundra environment of the western Arctic. Canadian Geotechnical Journal **7:** 420-432.

Mackenzie Gas Project. 2003. Mackenzie Gas Project. http://www.mackenziegasproject.com/index.asp Imperial Oil Limited, Toronto, Canada.

McCune, B. and J. B. Grace. 2002. Analysis of Ecological Communities. MjM Software Design, Durham, NC.

McCune, B. and M. J. Mefford. 2006. PC-ORD. Multivariate Analysis of Ecological Data. MjM software, Gleneden Beach, Oregon.

McKendrick, J. 2000. Vegetative responses to disturbance. Pages 35-56 *in* J. C. Truett and S. R. Johnson, editors. The Natural History of an Arctic Oil Field Academic Press, San Diego, CA.

McKendrick, J. D. 1987. Plant succession on disturbed sties, North slope, Alaska, U.S.A. Arctic and alpine research **19**:554-565.

McKendrick, J. D. 1991. Arctic tundra rehabilitation observations of progress and benefits to Alaska, USA. Agroborealis **23**:29-40.

McKendrick, J. D. 1996. Rehabilitating arctic tundra in Alaska. North American Water and Environment Congress & Destructive Water:764-769.

McKendrick, J. D. 1997. Long-term tundra recovery in Northern Alaska. Pages 503-518 *in* R. M. M. Crawford, editor. Disturbance and Recovery in Arctic Lands. Kluwer Academic Publishers, Netherlands.

McKendrick, J. D., R. C. Wilkinson, and R. G. B. Senner. 1997. Tundra plant succession and vascular plant species diversity. Agroborealis **29**:28-30.

McLaren, J. R. 2006. Effects of plant functional groups on vegetation dynamics and ecosystem properties. Arctic **59**:449-452.

Mielke, P. W. and K. J. Berry. 2007. Permutation Methods: A distance functional approach. 2 edition. Spring Series in Statistics, New York, NY.

Morse, P. D., C. R. Burn, and S. V. Kokelj. 2009. Near-surface ground-ice distribution, Kendall Island Bird Sanctuary, Western Arctic Coast, Canada. Permafrost and Periglacial Processes **20**:155-171.

Parks Canada. 1992. Appendix E - Management of Migratory Bird Sanctuaries in the Inuvialuit Settlement Region. http://www.pc.gc.ca/eng/pn-np/nt/aulavik/docs/plan2/sec13/page4.aspx. Canada Wildlife Service, Yellowknife, NT.

Pillai, K. C. S. 1955. Some new test criteria in multivariate analysis. The Annuals of Mathematical Statistics **26**:117-121.

Porsid, A. E. and W. J. Cody. 1980. Vascular Plants of Continental Northwest Territories, Canada National Museums of Canada, Ottawa, ON.

Quinn, G. P. and M. J. Keough. 2002. Experimental Design and Data Analysis for Biologists. Cambridge University Press, New York, NY.

R Development Core Team. 2009. R: A language and environment for statistical computing. R Foundation for Statistical computing, Vienna, Austria.

Rausch, J. and G. Kershaw. 2007. Short-term revegetation performance on gravel-dominated, human-induced disturbances, Churchill, Manitoba, Canada. Arctic, Antarctic, and Alpine Research **39**:16-24.

Raynolds, M. K. and D. A. Walker. 2009. Effects of deglaciation on circumpolar distribution of arctic vegetation. Canadian Journal of Remote Sensing **35**:118-129.

Schindler, D. W. and J. P. Smol. 2006. Cumulative effects of climate warming and other human activities on freshwaters of Arctic and subarctic North America. Pages 160-168. Royal Swedish Academy of Sciences.

Serreze, M. C., J. E. Walsh, F. S. Chapin, III, T. Osterkamp, M. Dyurgerov, V. Romanovsky, W. C. Oechel, J. Morison, T. Zhang, and R. G. Barry. 2000. Observational evidence of recent change in the northern high-latitude environment. Climatic Change **46**:159-207.

Serverson-Baker, C. 2004. Exploration and production drilling: A primer. Pembina Institute for Appropriate Development:1-50.

Shirazi, M. A., P. K. Haggerty, C. W. Hendricks, and M. Reporter. 1998. The role of thermal regime in tundra plant community restoration. Restoration Ecology **6**:111-117.

Simpson, E. H. 1949. Measurement of diversity. Nature **163**:688.

Smith, D. W. and T. D. W. James. 1985. Sump studies V: Ecological changes adjacent to sumps at exploratory well sites in the Mackenzie Delta and Northern Yukon. Northern Affairs Program.

Smith, M. W. 1975. Microclimatic influences on ground temperatures and permafrost distribution, Mackenzie Delta, Northwest Territories. Canadian Journal of Earth Sciences **12**:1421-1438.

SPSS Inc. 2009. SPSS Statistics. Chicago, Illinois.

Srivastava, D. S. and R. L. Jefferies. 1995. The effect of salinity on the leaf and shoot demography of two arctic forage species. Journal of Ecology **83**:421-430.

Streever, W. J., J. McKendrick, L. Fanter, S. C. Anderson, J. Kidd, and K. M. Portier. 2003. Evaluation of percent cover requirements for revegetation of disturbed sites on Alaska's North slope. Arctic **56**:234-248.

Sturm, M., J. P. McFadden, G. E. Liston, F. S. Chapin, III, C. H. Racine, and J. Holmgren. 2001. Snow–shrub interactions in arctic tundra: A hypothesis with climatic implications. Journal of Climate **14**:336-344.

Sturm, M., J. Schimel, G. Michaelson, J. M. Welker, S. F. Oberbauer, G. E. Liston, J. Fahnestock, and V. E. Romanovsky. 2005. Winter biological processes could help convert arctic tundra to shrubland. Bioscience **55**:17-26.

Svoboda, J. and G. H. R. Henry. 1987. Succession in marginal arctic environments. Arctic and alpine research **19**:373-384.

Tape, K., M. Sturm, and C. H. Racine. 2006. The evidence for shrub expansion in Northern Alaska and the Pan-Arctic. Global Change Biology **12**:686-702.

Trimble Navigation Limited. 2007. Trimble Buisness Center. Dayton, Ohio.

Truett, J. C. and S. R. Johnson. 2000. The Natural History of an Arctic Oil Field: Development and the Biota. Academic Press, San Diego, CA.

Truett, J. C. and K. Kertell. 1992. Tundra disturbance and ecosystem production: Implications for impact assessment. Environmental management **16**:485-949.

Urbanska, K. M. 1997. Safe sites - interface of plant population ecology and restoration ecology. Pages 81-110 *in* K. M. Urbanska, N. R. Webb, and E. J. Edwards, editors. Restoration Ecology and Sustainable Development. Cambridge University Press, Cambridge, UK.

Walker, D. A. and M. D. Walker. 1991. History and pattern of disturbance in Alaskan Arctic terrestrial ecosystems: A hierarchical approach to analyzing landscape change. Journal of Applied Ecology **28**:244-276.

Walker, M. D., C. H. Wahren, R. D. Hollister, G. H. R. Henry, L. E. Ahlquist, J. M. Alatalo, M. S. Bret-Harte, M. P. Calef, T. V. Callaghan, A. B. Carroll, H. E. Epstein, I. S. Jonsdottir, J. A. Klein, B. Magnusson, U. Molau, S. F. Oberbauer, S. P. Rewa, C. H. Robinson, G. R. Shaver, K. N. Suding, C. C. Thompson, A. Tolvanen, O. Totland, P. L. Turner, C. E. Tweedie, P. J. Webber, and P. A. Wookey. 2006. Plant community responses to experimental warming across the tundra biome. Proceedings of the National Academy of Sciences of the United States of America **103**:1342-1346.

Wood, S. N. 2006. Generalized Additive Models: An Introduction with R. Chapman & Hall/CRC, Boca Raton, FL.

Wood, S. N. 2008. Fast stable direct fitting and smoothness selection for generalized additive models. Journal of the Royal Statistical Society. Series A. Statistics in society **70**:495-524.

Wright, S. J. 2008. A Revegetation Manual for Alaska.*in* P. Hunt, editor. Alaska Plant Material Center, Alaska Department of Natural Resources, Palmer, AK.

Younkin, W. E. and H. E. Martens. 1976. Progress report on rig site seeding tests in the Mackenzie Delta Region, NWT.*in* W. Younkin, editor. Revegetation Studies in the Northern Mackenzie River valley region. Prepared by Northern Engineering Services Company Ltd. for Canadian Arctic Gas Study Ltd.

Younkin, W. E. and H. E. Martens. 1987. Long-term success of seeded species and their influence on native species invasion at abandoned rig site A-01 Caribou Hills, NWT, Canada. Arctic and alpine research **3**:566-571.

Zar, J. H. 1999. Biostatistical Analysis. Prentice Hall, Upper Saddle River, NJ.